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IMPACT OF DISTURBANCE REGIMES ON COMMUNITY AND LANDSCAPE
BIODIVERSITY IN ATLANTIC COASTAL PINE BARREN ECOREGION
STREAMS

A Dissertation Presented
by
SEAN T McCANTY

Submitted to the Office of Graduate Studies,
University of Massachusetts Boston,
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

AUGUST 2020

ENVIRONMENTAL BIOLOGY PROGRAM

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IMPACT OF DISTURBANCE REGIMES ON COMMUNITY AND LANDSCAPE
BIODIVERSITY IN ATLANTIC COASTAL PINE BARREN ECOREGION
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SEAN T. McCANTY

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ABSTRACT

IMPACT OF DISTURBANCE REGIMES ON COMMUNITY AND LANDSCAPE BIODIVERSITY IN ATLANTIC COASTAL PINE BARREN ECOREGION STREAMS

AUGUST 2020

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Streams are dynamic systems shaped by geographic location, hydrology, riparian vegetation, and in-stream habitat. Furthermore, ecosystem disturbance plays a major role in structuring stream communities and ecosystem processes. Disturbances include natural occurrences, such as flooding, drought, and fire events and anthropogenic disturbances such as land use changes, damming, and pollution. Agricultural use acts as a press disturbance regime, homogenizing the surrounding landscape and simplifying in-stream habitat, leaving legacy effects after farming ceases. Active restoration is intended to ameliorate these effects by reintroducing variation, with the goal of shifting the ecosystem into a more diverse and natural state. The act of restoration therefore acts as a pulse disturbance, attempting to shift the community from one system state to another through habitat and process alteration. Active restoration of an in-fallow (since 2010) flow-through cranberry bog in Southeastern Massachusetts occurred in late 2015, allowing the ability to set up a Before-After-Control-Impact (BACI) design to investigate the effects of

restoration's disturbance phase. Structural, compositional, and functional attributes of macroinvertebrate and habitat/ecosystem biodiversity were evaluated over a 3-year study at the restored site, with an active flow-through cranberry bog and a least-impacted stream acting as regional controls of high and low disturbance respectively. As expected, we saw compositional shifts in the macroinvertebrate assemblage and several measures of ecosystem function related to the perturbation of disturbance, but limited evidence of long-term shift away from initial conditions following restoration.

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CHAPTER 1

INTRODUCTION

Ecosystems are inherently dynamic, with structures and processes driven by changes to the physical environment, community-wide interactions, and species movement (Folke et al., 2004). The concept of a single climax community, or the composition of the community that would exist in the absence of perturbation, is repeatedly challenged in favor of multiple stable states established by different pressures and attractors (Folke et al., 2004; Holling, 1996). Thus, even a system deemed at equilibrium experiences stochastic effects of temperature, weather, and extreme events like fire, yet small perturbations are usually insufficient to reach a tipping point to a new equilibrium (Folke et al., 2004; Tilman, 1999).

Stability of ecosystem states relates to the concepts of resistance and resilience, where the former refers to a system's ability to endure perturbation with no biological change and the latter refers to the speed with which system recovers from any change post-disturbance (Gunderson, 2000; Holling, 1996; Nimmo et al., 2015). Although clear-cut in theory, distinguishing resistance and resilience in systems under either constant pressure or frequent disturbances becomes difficult (Nimmo et al., 2015). This is compounded when

a system may have many potential states that are not well studied, and resilience may thus be conflated with transitions between states (Gunderson, 2000).

Ecological disturbances beyond background stochasticity are generally characterized as press, pulse, or ramp disturbances, and are based on the intensity, frequency, and duration of the perturbation (Lake, 2000). Press disturbances are constant stressors to the system that do not vary significantly in intensity, such as consistent water withdrawal from a stream (Lake, 2000). Pulse disturbances are temporally relatively short, such as flooding events; disrupting the ecosystem and then allow for recovery or change (Lake, 2000). Finally, ramp disturbances are exemplified by droughts, where the perturbation increases in intensity over time (Lake, 2000, 2003). All forms of disturbance are followed by a community response, usually a change in some measure of productivity and a variable response in terms of biodiversity (Lake, 2000). The Intermediate Disturbance Hypothesis (IDH), for example, states that biodiversity of sessile organisms may benefit from some moderate level of disturbance intensity and/or frequency as competitive exclusion is prevented (Connell, 1978). Biodiversity maximization during intermediate, regular disturbances is recognized in the context of regenerative fires (Templeton et al., 2011) and increased stream food web length (Power et al., 1995). However, other studies have called the intermediate disturbance hypothesis into question – for example, in cases of system-specific productivity decline and negative empirical results (Svensson et al., 2012). For aquatic systems, the interplay of abundance, richness, and evenness can result in divergence from IDH in favor of dynamic-equilibrium predictions by lowering overall abundance but increasing species richness at highest disturbance levels (McCabe and Gotelli, 2000).

In general, various forms of disturbance shape freshwater streams, including periodic flooding, seasonal drought, decreased flow, and temperature fluctuations (Lake, 2003; Power et al., 2013). Such variation in abiotic conditions is recognized as structuring the biological community through direct (e.g. habitat changes) and indirect (e.g. biotic interactions) mechanisms (Stanley et. al 2010, Power et. al 2013). The necessity of certain disturbance mechanisms in understanding stream dynamics is codified in stream structuring concepts such as the flood-pulse model (Junk et al., 1989) and riverine landscape ecology (Wiens, 2002), as well as underlying models of stream community assembly (Poff, 1997). These add to the fundamental understanding that stream biota is shaped largely by allochthonous and autochthonous energy sources; channel geomorphology and physical habitat; the riparian environment and the watershed; and connectivity from headwaters to outflow as described in the River Continuum Concept (RCC) (Fausch et al., 2002; Vannote et al., 1980). In addition to the biota themselves, trophic interactions also are shaped by riverine dynamics (Woodward and Hildrew, 2002). The importance of scale, both spatially and temporally, becomes clear as streams begin as small headwaters and join to form river system networks, with different driving processes (Frissell et al., 1986) and forms of disturbance (Stanley et al., 2010) becoming more important at higher stream orders.

While natural disturbance can create a diverse array of microhabitats within a stream, disturbances/perturbations also have become common terms to describe any human or natural stressor on streams (Formann et al., 2013; Stanley et al., 2010). Indeed, for aquatic systems, press disturbances are largely the result of human activities, especially altered land-uses (Lake, 2000). Urbanization is a common degrader of stream integrity and

biodiversity, leading to the so-called urban stream syndrome, which acts as a catch-all term for the synergistic effects of co-occurring stream impacts (Walsh et al., 2005). Agriculture also results in habitat simplification and sediment loading consistent with press disturbances (Maloney and Weller, 2011; Waite, 2013). In particular, systems using or directly bordering streams magnify this impact and may impact the system through channelization of the stream bed and simplifying the in-stream habitat and flow regime (Tullos et al., 2009).

Due to the stochastic nature of disturbances, and particularly pulse disturbances, studies are often limited by either the absence or only minimal presence of pre-disturbance data and reliable establishment of baseline conditions. Attempts to substitute space for time in these assessments may result in confounding variables and site specific influences (Pickett, 1989). Land-use alterations, such as restorations, then provide a solid avenue for known disturbance, allowing the collection of data before, during, and after implementation of a press or pulse disturbance.

Disparate methods of quantifying stream disturbance recoveries make larger scale meta-analysis difficult. Often, studies only incorporate measures directly related to the disturbance impact or the major component stressor to the system (Palmer et al., 2005). Examples in the ecological literature, however, make clear that indirect effects result from altering a disturbance regime, such as the changes in the dispersal of lizards in the face of fire suppression or prescribed burns (Templeton et al., 2011). In the context of evaluating system response to disturbance then, a comprehensive framework that is sensitive to both biotic and abiotic changes is required.

Biodiversity is a complex composite, more than simply the sum of diversity measures from the level of the gene, to the population, to the community level, and to the ecosystem (Noss, 1990; Tilman, 1999). In addition to the hierarchy of scale, biodiversity consists of compositional, structural, and functional attributes (Noss, 1990; Redford and Richter, 1999), allowing for a finer scale of assessing change. For example, invertebrate assemblages may vary compositionally due to landscape filtering rather than local scale effects (Poff, 1997) while human activities such as impoundment may create local scale changes to structural – such as decreasing submerged snag density and occupying range (Scholl et al., 2015) – or functional aspects – such as a relative decline in non-grazer feeding group densities (Rabeni et al., 2005; Wallace and Webster, 1996). By working with a biodiversity definition that operates both at varying spatial scales and differentiating between composition, structure, and function, disturbances can be more accurately described in terms of their biological and ecosystem impact.

Common diversity measures have varying responses to disturbance under different regimes, including divergent predictions for richness and evenness along a disturbance and productivity gradient (Svensson et al., 2012). In contrast, biodiversity analysis allows a system to be examined across the multiple dimensions through which a disturbance can act. Indeed, some measures of α -diversity, such as Shannon's diversity, may be less sensitive to ecosystem wide alterations that can be detected by analysis of food web dynamics (Christianen et al., 2016). Using biodiversity as a lens for examining response to disturbance allows the direct changes to ecosystem composition, function, and structure to be mapped to community level attribute changes.

STATEMENT OF THE PROBLEM

Modification of the stream system and surrounding landscape often leads to habitat loss with biodiversity declines. Land use change and habitat loss are recognized as leading global drivers of biodiversity change, and the most significant ones for the future of streams and rivers (Sala et al., 2000). Since freshwater biodiversity is being lost at far greater rates than terrestrial or marine biodiversity, attempts to slow these drivers is critical to preserving this vital resource (Sala et al., 2000). To combat this, restoration or remediation of impaired landscapes has been occurring with increasing frequency (BenDor et al., 2015; Palmer et al., 2010). Broadly speaking, these activities can range in goals and specific activities, such as the replanting of native vegetation (Harrington, 1999), habitat replacement for a single species (Dumke et al., 2010), the removal of an impoundment (Tullos et al., 2009) or remediation of a contaminant (Adams et al., 2005).

Even when such a land-use based press disturbance is removed, such as through the cessation of farming, legacy affects may linger (Quist and Schultz, 2014). Channelization, sediment deposition, and other forms of channel geomorphology changes and habitat simplification remain and continue to impact the system (Quist and Schultz, 2014). In a study of the legacy of passive restoration of flow-through cranberry bogs, invertebrate diversity was correlated with time spent in restoration, yet differences were still found between sites that had never been farmed and those in restoration for decades (Engel, 2010). Similar studies on passive restoration, or the cessation of harmful activities, have shown it an insufficient substitute for recreating natural habitat conditions (Kauffman et al., 1997; Kristensen et al., 2013). Attempts to more actively remediate or restore such

degraded systems, and in particular stream restoration activities, have been attempted for the past several decades, often with varying goals including both social and ecological (Smith et al., 2016). Due to the expense of these projects, effort is often undertaken at the local, or “reach,” scale. As a result, the ecological outcome is often unclear, due to limited investment in monitoring and the confounding factors of larger watershed impact (Bernhardt and Palmer, 2011; Palmer et al., 2005). Reach restoration activities that have been touted as more successful follow a process-based approach, where the goal of the restoration is to reestablish broader functions or to re-engineer structural diversity and environmental complexity, such as channel sinuosity and natural flow variability, rather than locale-specific end-points (Wohl et al., 2005).

From the perspective of the stream channel, these active restorations constitute a pulse disturbance, as the stream experiences direct impacts through widening, creation of new channel, and other alterations that would naturally occur with restructuring pulse disturbances such as flooding (Lake, 2003; Stanley et al., 2010; Thomas et al., 2015). Examples across restoration or remediation literature tend to reinforce the “restoration as disturbance” paradigm. In particular, remediation of contaminated sediment generally includes an initial re-suspension followed by redevelopment of natural disturbance regimes (Harper and Peckarsky, 2005). Small dam removal also can be modeled as a pulse disturbance, where initial sediment loading gives way to a more varied flow regime (Tullos et al., 2014). The goal of such a directed pulse is the reduction of legacy impacts that would take centuries of natural perturbation to achieve.

In Massachusetts, as well as throughout the Northeast, there has been a greater decline in agriculture than in any other land use since 1971 (decreased by 17.6%; Bureau

of Geographic Information, 2016). Although historic changes in land use in the Northeast have been associated with forest regrowth far more homogenous than pre-agriculture, these observations are based largely on abandoned plots, substituting the removal of agricultural disturbance with maintained changes to low-intensity disturbance associated with human activities (Foster et al., 1998). Better study of these systems under active restoration can elucidate the changes experienced across all levels and attributes of biodiversity.

Flow-through cranberry bogs are an example of high impact agriculture, since the demands of farming involve regulating flow and flooding and direct channel manipulation on top of input of excess nutrients, pesticides, and sediment. Additionally, given the history of cranberry agriculture in Southern Massachusetts, many of these flow-through bogs have been in use for centuries (Cape Cod Cranberry Growers Association, 2016). As a result, the associated stream ecosystems have been under continual press disturbance, albeit with within-year variation occurring throughout, and the trajectory the community and ecosystem processes would take when released from such a press disturbance remains an open question.

Restoration of a flow through cranberry bog, with concurrent study of conservation land and an actively used flow through cranberry bog allows me to examine the influence of pulse disturbance, background disturbance, and press disturbance respectively. The ability to have data before the restoration allows us to better understand the changes occurring and predict trajectories for changes the system may experience.

GENERAL EXPERIMENTAL DESIGN AND STUDY AREAS

Research was conducted at three (3) treatment sites, representing directed pulse disturbance (Cranberry Bog Restoration, Tidmarsh Farms, CBR), background disturbance (Least impacted reference, Mashpee River, LIR), and an agricultural press disturbance (Active Cranberry Bog, ACB) (Figure 1). All three (3) treatments were low-order (1st-3rd) streams fed by primarily groundwater and drainage fed ponds in the Atlantic Coastal Pine Barren Ecoregion. Additionally, these watersheds are similar in composition of land use/land cover and discharge (Table 1). Each treatment site consisted of three (3) replicate stations defined as a 100m length reach where data will be collected. Each station was separated by at least 100m in river channel length. All treatments were sampled seasonally (i.e. spring, summer, and autumn) starting in 2015, 1 year prior to the pulse disturbance at the CBR treatment, and continued for 2 years post-pulse disturbance at the CBR treatment. With the exception of habitat and seasonal changes to water column chemistry, this document investigates primarily changes related to summer sample collections.

The CBR treatment site, Tidmarsh Farms, is a former flow-through cranberry bog consisting of 192 acres. The farm obtains water from Beaver Dam Brook, fed by Beaver Dam Pond and Fresh Pond, both primarily groundwater fed retention ponds. Cranberry farming at CBR was decommissioned in 2010, when farming ceased and water controls were removed. This marked the beginning of passive restoration (Kauffman et al., 1997), allowing the landscape to variation before proceeding with a 6-month active restoration in late Autumn 2015. This consisted of landscape modification in the form of a new sinuous channel being dredged, adding large woody debris, creating macrohabitat sequences,

revegetating the riparian zone, and raising the water table to reconnect the flood plain and create floodplain wetland features including fens, wet meadows, and scrub swamps. The sum of these alterations to the landscape can be considered an ecosystem pulse disturbance, greatly altering the abiotic environment over a short time frame, followed by ecological recovery. The Christian Lab has been involved in assessing change in macroinvertebrate, water chemistry, and in-stream habitat at this site since 2010 (Franck, 2017; O’Brion, 2012). Three (3) stations were selected from pre-existing stations after data demonstrated no significant across-site variation for low-order stream sites (Franck, 2017). Following restoration and subsequent rechannelization, new stations were established along the new channel with approximately the same geographic separation.

The LIR treatment site, Mashpee River, near Mashpee, MA drains the Mashpee Pond into Buzzard Bay in southern Massachusetts. The sampled portion of the stream falls within forested conservation land managed by the Mashpee River Reservation (248 acres) since 1959 and is part of the Trustees of Reservations network (Trustees of the Reservations, 2017). The river and directly adjacent land have no history of agricultural use. The three (3) LIR treatment stations were distributed along the length of the river within the conservation land to minimize autocorrelative effects.

The ACB treatment site was a flow-through bog of approximately 30 acres in Plymouth County, MA (Owner identifying information is withheld as part of a non-disclosure agreement). It is pond-fed, and water level and flow is controlled at several points along the property. The channel has no canopy cover and is maintained for cranberry growth and harvest, including periodic dredging and removal of submerged aquatic vegetation. The three (3) ACB site stations were spaced along the property to maximize

distance between the stations and place each within its own agricultural cell. Active growth and harvest of cranberries occurred in both 2015 and 2016, however in 2017 the acreage was left fallow due to owner production considerations.

CHAPTER DELINEATION

The organization of my dissertation is modeled after the multiple monography style outlined by the Office of Graduate Studies at the University of Massachusetts Boston. This organization includes an introductory (background and literature review) chapter; three stand-alone manuscript-style research chapters, including full titles, abstracts, introduction, methods, results, discussion, literature cited, and illustrations; and an overall dissertation conclusion chapter. The following paragraphs provide a brief summary of the three stand-alone manuscript-style research chapters.

Chapter 2 examines watershed scale land use and reach scale physical habitat changes across treatments and time. Active restoration provides the opportunity for rapid changes to both the in-stream habitat, through rechannelization and introduced heterogeneity, as well as restructuring a significant portion of the watershed land use through riparian alterations. Assessment of both together reveals the scope of active over passive restoration and the potential for colonization by previously absent taxa.

In Chapter 3, the compositional and functional aspects of community biodiversity is determined by examining freshwater macroinvertebrate (FMI) assemblages. By examining the FMI assemblage changes through taxonomic and functional composition, early shifts and potential trends in restoration can be identified, without the ambiguity

interpreting simple diversity metrics may present. Early community changes following restoration, especially when the pre-restoration community is well studied, remain understudied and provide the ability of determining longer term trends in system recovery.

Finally, Chapter 4 investigates the ecosystem level compositional, structural, and functional aspects of biodiversity by analysis of the stream processes. In particular, changes in water column nutrient concentrations and assessment of nutrient limitation address alterations made during rechannelization and the reconnection to groundwater and submerged peat layer. Meanwhile, variations in stream metabolism and functional metrics of in-stream food webs provides broad senses of shifts in biotic interactions which may not be reflected in compositional assessments alone.

Restoration's role as a perturbation to the system is well established in stream systems (Stanley et al., 2010). However, early changes following restoration can determine whether the active process takes the trajectory of a pulse disturbance without alteration of ecosystem state, or whether trends towards an alternative state are apparent. At the crux of this study, the integration of habitat, community, and matter and energy dynamics provide a weight-of-evidence approach to interpreting changes with limited replication spatially and temporally. Additionally, the framework for investigation allows for potential follow-up study and a broader investigation of the co-variability of restoration changes.

TABLES

Table 1: Watershed and reach scale values for all 3 treatments. Basin scale values were computed on drainage areas obtained through StreamStats (United States Geological Survey, 2016) and datalayers from MassGIS (Bureau of Geographic Information, 2016). Reach scale values are based on Summer 2015 values, representing summer base flow conditions.

Treatment	ACB	CBR	LIR
Drainage Area (km ²)	18.24	8.21	44.71
Site Elevation (m)	70	40	10-20
% Forest	74.94	50.51	56.75
% Non-Cranberry Agriculture	0.59	0.84	1.63
% Cranberry Agriculture	1.9	9.89	0.12
% Urban	4.83	29.8	17.82
Avg. Discharge (m ³ /s)	0.325	0.226	0.175
Avg. Stream Width (m)	5.6	7.6	6.7
Avg. Depth (m)	0.50	0.28	0.14
Avg. Substrate	Sand	Sand/Fine	Gravel
Riparian Description	Cranberry/ Mowed Grass	Cranberry/ Shrubs/Grasses	Forested

FIGURES

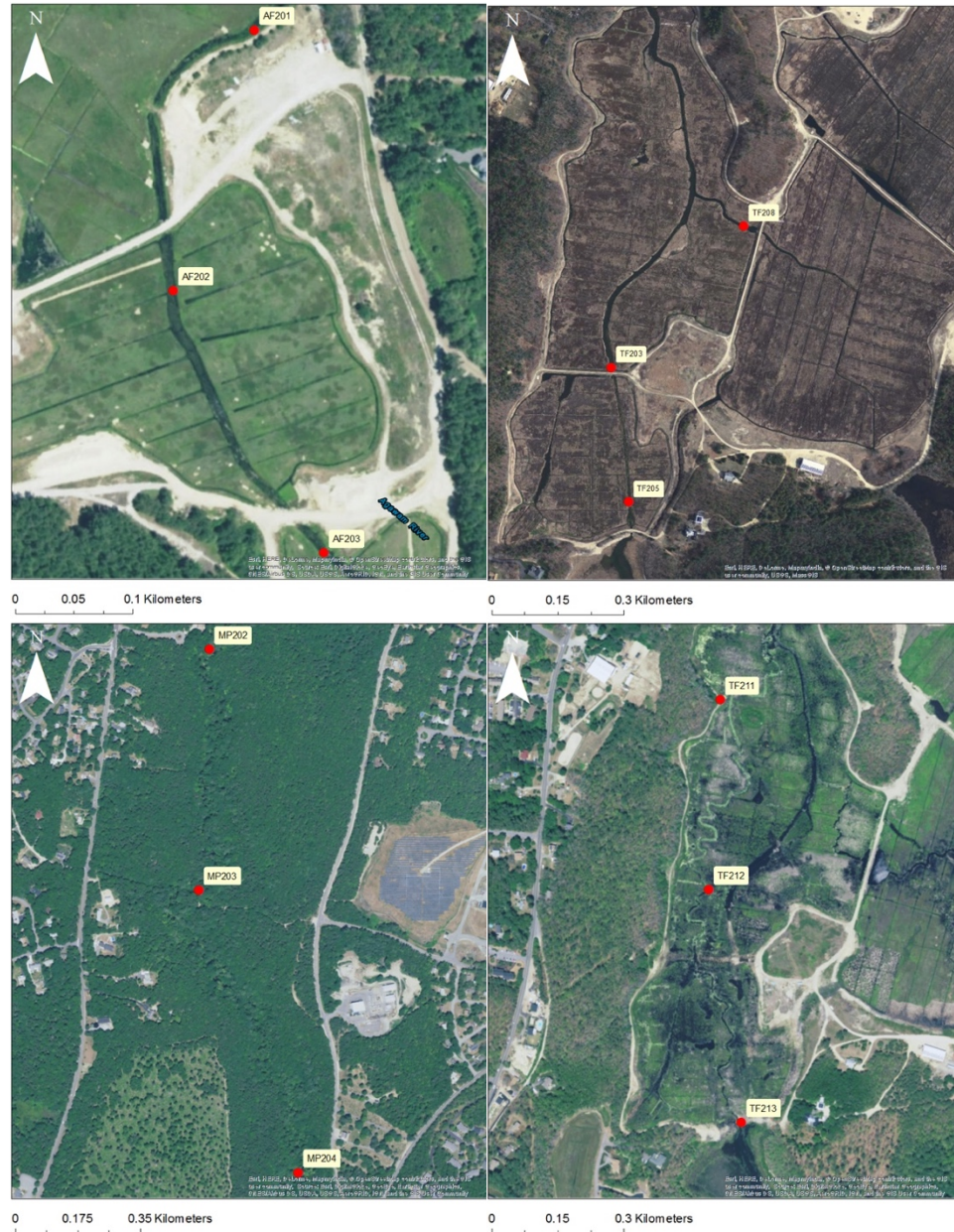


Figure 1: Study locations. From top left proceeding clockwise: Active Cranberry Bog (ACB), Cranberry Bog Restoration (CBR) in 2015, Cranberry Bog Restoration (CBR) in 2016, and the Least Impacted Reference (LIR) treatments. All station markers mark the upstream end of the 100m reach. The two separate maps for CBR represent sampling locations pre- and post-active restoration as stations had to be re-established to remain within the stream channel. Map made with ArcGIS (ESRI) by S. McCanty.

LITERATURE CITED

- Adams, S.M., Ryon, M.G., and Smith, J.G. (2005). Recovery in diversity of fish and invertebrate communities following remediation of a polluted stream: investigating causal relationships. *Hydrobiologia* 542, 77–93.
- BenDor, T., Lester, T.W., Livengood, A., Davis, A., and Yonavjak, L. (2015). Estimating the Size and Impact of the Ecological Restoration Economy. *PLOS ONE* 10, e0128339.
- Bernhardt, E.S., and Palmer, M.A. (2011). River restoration: the fuzzy logic of repairing reaches to reverse catchment scale degradation. *Ecol. Appl.* 21, 1926–1931.
- Bureau of Geographic Information (2016). Bureau of Geographic Information (MassGIS) (Commonwealth of Massachusetts, Executive Office of Technology and Security Services).
- Cape Cod Cranberry Growers Association (2016). History of Cranberries.
- Christianen, M., van der Heide, T., Holthuijsen, S., van der Reijden, K., Borst, A., and Olff, H. (2016). Biodiversity and food web indicators of community recovery in intertidal shellfish reefs. *Biol. Conserv.*
- Connell, J.H. (1978). Diversity in Tropical Rain Forests and Coral Reefs. *Science* 199, 1302–1310.
- Dumke, J.D., Hrabik, T.R., Brady, V.J., Gran, K.B., Regal, R.R., and Seider, M.J. (2010). Channel Morphology Response to Selective Wood Removals in a Sand-Laden Wisconsin Trout Stream. *North Am. J. Fish. Manag.* 30, 776–790.
- Engel, H. (2010). Stream quality among active and restoring river-based cranberry bogs. Unpubl. MBL Woods Hole.
- ESRI ArcGIS 10.4.
- Fausch, K.D., Torgersen, C.E., Baxter, C.V., and Li, H.W. (2002). Landscapes to Riverscapes: Bridging the Gap between Research and Conservation of Stream Fishes A Continuous View of the River is Needed to Understand How Processes Interacting among Scales Set the Context for Stream Fishes and Their Habitat. *BioScience* 52, 483–498.
- Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., and Holling, C.S. (2004). Regime Shifts, Resilience, and Biodiversity in Ecosystem Management. *Annu. Rev. Ecol. Evol. Syst.* 35, 557–581.

- Formann, E., Egger, G., Hauer, C., and Habersack, H. (2013). Dynamic disturbance regime approach in river restoration: concept development and application. *Landsc. Ecol. Eng.* 10, 323–337.
- Foster, D.R., Motzkin, G., and Slater, B. (1998). Land-Use History as Long-Term Broad-Scale Disturbance: Regional Forest Dynamics in Central New England. *Ecosystems* 1, 96–119.
- Franck, E. (2017). Establishment of Baseline Physical, Chemical, and Biological Stream Conditions at a Passive Cranberry Bog Restoration. Grad. Masters Theses.
- Frissell, C.A., Liss, W.J., Warren, C.E., and Hurley, M.D. (1986). A hierarchical framework for stream habitat classification: Viewing streams in a watershed context. *Environ. Manage.* 10, 199–214.
- Gunderson, L.H. (2000). Ecological resilience - in theory and application. *Annu. Rev. Ecol. Syst.* 31, 425–439.
- Harper, M.P., and Peckarsky, B.L. (2005). Effects of pulsed and pressed disturbances on the benthic invertebrate community following a coal spill in a small stream in northeastern USA. *Hydrobiologia* 544, 241–247.
- Harrington, C.A. (1999). Forests planted for ecosystem restoration or conservation. In *Planted Forests: Contributions to the Quest for Sustainable Societies*, J.R. Boyle, J.K. Winjum, K. Kavanagh, and E.C. Jensen, eds. (Springer Netherlands), pp. 175–190.
- Holling, C.S. (1996). Surprise for Science, Resilience for Ecosystems, and Incentives for People. *Ecol. Appl.* 6, 733–735.
- Junk, W.J., Bayley, P.B., and Sparks, R.E. (1989). The flood pulse concept in river-floodplain systems. In *Proceedings of the International Large River Symposium*, D.P. Dodge, ed. pp. 110–127.
- Kauffman, J.B., Beschta, R.L., Otting, N., and Lytjen, D. (1997). An Ecological Perspective of Riparian and Stream Restoration in the Western United States. *Fisheries* 22, 12–24.
- Kristensen, E.A., Thodsen, H., Dehli, B., Kolbe, P.E.Q., Glismand, L., and Kronvang, B. (2013). Comparison of active and passive stream restoration: effects on the physical habitats. *Geogr. Tidsskr.-Dan. J. Geogr.* 113, 109–120.
- Lake, P.S. (2000). Disturbance, patchiness, and diversity in streams. *J. North Am. Benthol. Soc.* 19, 573–592.
- Lake, P.S. (2003). Ecological effects of perturbation by drought in flowing waters. *Freshw. Biol.* 48, 1161–1172.

- Maloney, K.O., and Weller, D.E. (2011). Anthropogenic disturbance and streams: land use and land-use change affect stream ecosystems via multiple pathways. *Freshw. Biol.* *56*, 611–626.
- McCabe, D.J., and Gotelli, N.J. (2000). Effects of disturbance frequency, intensity, and area on assemblages of stream macroinvertebrates. *Oecologia* *124*, 270–279.
- Nimmo, D.G., Mac Nally, R., Cunningham, S.C., Haslem, A., and Bennett, A.F. (2015). Vive la résistance: reviving resistance for 21st century conservation. *Trends Ecol. Evol.* *30*, 516–523.
- Noss, R.F. (1990). Indicators for Monitoring Biodiversity: A Hierarchical Approach. *Conserv. Biol.* *4*, 355–364.
- O’Brion, K. (2012). Physical and Biological Assessment of the Eel River Headwaters Restoration Site in Plymouth, Massachusetts. Grad. Masters Theses.
- Palmer, M. a., Bernhardt, E. s., Allan, J.D., Lake, P. s., Alexander, G., Brooks, S., Carr, J., Clayton, S., Dahm, C.N., Follstad Shah, J., et al. (2005). Standards for ecologically successful river restoration. *J. Appl. Ecol.* *42*, 208–217.
- Palmer, M.A., Menninger, H.L., and Bernhardt, E. (2010). River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? *Freshw. Biol.* *55*, 205–222.
- Pickett, S.T.A. (1989). Space-for-Time Substitution as an Alternative to Long-Term Studies. In *Long-Term Studies in Ecology*, G.E. Likens, ed. (Springer New York), pp. 110–135.
- Poff, N.L. (1997). Landscape Filters and Species Traits: Towards Mechanistic Understanding and Prediction in Stream Ecology. *J. North Am. Benthol. Soc.* *16*, 391–409.
- Power, M.E., Parker, M.S., and Wooten, J.T. (1995). Disturbance and food chain length in rivers. In *Food Webs: Integration of Patterns and Dynamics*, G.A. Polis, and K.O. Winemiller, eds. (New York: Chapman Hall), p.
- Power, M.E., Holomuzki, J.R., and Lowe, R.L. (2013). Food webs in Mediterranean rivers. *Hydrobiologia* *719*, 119–136.
- Quist, M., and Schultz, R. (2014). Effects of Management Legacies on Stream Fish and Aquatic Benthic Macroinvertebrate Assemblages. *Environ. Manage.* *54*, 449–464 16p.
- Rabení, C.F., Doisy, K.E., and Zweig, L.D. (2005). Stream invertebrate community functional responses to deposited sediment. *Aquat. Sci.* *67*, 395–402.

- Redford, K.H., and Richter, B.D. (1999). Conservation of Biodiversity in a World of Use. *Conserv. Biol.* 13, 1246–1256.
- Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., et al. (2000). Biodiversity - Global biodiversity scenarios for the year 2100. *Science* 287, 1770–1774.
- Scholl, E.A., Rantala, H.M., Whiles, M.R., and Wilkerson, G.V. (2015). Influence of Flow on Community Structure and Production of Snag-Dwelling Macroinvertebrates in an Impaired Low-Gradient River. *River Res. Appl.* n/a-n/a.
- Smith, R.F., Hawley, R.J., Neale, M.W., Vietz, G.J., Diaz-Pascacio, E., Herrmann, J., Lovell, A.C., Prescott, C., Rios-Touma, B., Smith, B., et al. (2016). Urban stream renovation: incorporating societal objectives to achieve ecological improvements. *Freshw. Sci.* 35, 364–379.
- Stanley, E.H., Powers, S.M., and Lottig, N.R. (2010). The evolving legacy of disturbance in stream ecology: concepts, contributions, and coming challenges. *J. North Am. Benthol. Soc.* 29, 67–83.
- Svensson, J.R., Lindegarth, M., Jonsson, P.R., and Pavia, H. (2012). Disturbance–diversity models: what do they really predict and how are they tested? *Proc R Soc B* rspb20112620.
- Templeton, A.R., Brazeal, H., and Neuwald, J.L. (2011). The transition from isolated patches to a metapopulation in the eastern collared lizard in response to prescribed fires. *Ecology* 92, 1736–1747.
- Thomas, G., Lorenz, A.W., Sundermann, A., Haase, P., Peter, A., and Stoll, S. (2015). Fish community responses and the temporal dynamics of recovery following river habitat restorations in Europe. *Freshw. Sci.* 34, 975–990.
- Tilman, D. (1999). The ecological consequences of changes in biodiversity: a search for general principles. *Ecology* 80, 1455–1474.
- Trustees of the Reservations (2017). Mashpee River Reservation.
- Tullos, D.D., Penrose, D.L., Jennings, G.D., and Cope, W.G. (2009). Analysis of functional traits in reconfigured channels: implications for the bioassessment and disturbance of river restoration. *J. North Am. Benthol. Soc.* 28, 80–92.
- Tullos, D.D., Finn, D.S., and Walter, C. (2014). Geomorphic and Ecological Disturbance and Recovery from Two Small Dams and Their Removal: e108091. *PLoS One San Franc.* 9, e108091.

- United States Geological Survey (2016). The Streamstats program, online at <http://streamstats.usgs.gov>.
- Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., and Cushing, C.E. (1980). The River Continuum Concept. *Can. J. Fish. Aquat. Sci.* 37, 130–137.
- Waite, I.R. (2013). Agricultural disturbance response models for invertebrate and algal metrics from streams at two spatial scales within the U.S. *Hydrobiologia* 726, 285–303.
- Wallace, J.B., and Webster, J.R. (1996). The role of macroinvertebrates in stream ecosystem function. *Annu. Rev. Entomol.* 41, 115–139.
- Walsh, C.J., Roy, A.H., Feminella, J.W., Cottingham, P.D., Groffman, P.M., and Morgan, R.P. (2005). The urban stream syndrome: current knowledge and the search for a cure. *J. North Am. Benthol. Soc.* 24, 706–723.
- Wiens, J.A. (2002). Riverine landscapes: taking landscape ecology into the water. *Freshw. Biol.* 47, 501–515.
- Wohl, E., Angermeier, P.L., Bledsoe, B., Kondolf, G.M., MacDonnell, L., Merritt, D.M., Palmer, M.A., Poff, N.L., and Tarboton, D. (2005). River restoration. *Water Resour. Res.* 41, W10301.
- Woodward, G., and Hildrew, A.G. (2002). Food web structure in riverine landscape. *Freshw. Biol.* 47, 777–798.

CHAPTER 2

CHANGES TO WATERSHED AND REACH SCALE HABITAT FEATURES ASSOCIATED WITH RESTORATION RELATED PULSE DISTURBANCE

ABSTRACT

Land use changes remain one of the leading global change drivers leading to biodiversity loss in terrestrial and aquatic systems. Development of “natural” (i.e. forested, grassland, or wetland) spaces and landscape fragmentation reduce local biodiversity through direct impacts to the water column and watershed scale fragmentation, which inhibits adult dispersal of aquatic insects. This case-study seeks to determine if the restoration of a former cranberry bog in Plymouth, MA has resulted in near-term measurable changes to the composition and fragmentation of watershed land use or local scale in-stream habitat diversity. A 3-year observational field study beginning one year prior to reconstruction was conducted at the restored cranberry bog and two control treatment sites: an active cranberry bog reference and a least impacted reference (i.e. has never been used for modern agriculture). Analysis of watershed level composition and fragmentation was done on pre- and post-restoration land use parameters. Seasonal inventories of in-stream habitat features including depth, substrate, macrohabitat, and in-stream cover were taken from 2015 to 2017. I found that post-restoration, watershed land use changed by roughly 10% in composition of natural land uses, the result of reclassification of land within the restoration

treatment. At the local scale however, there was no significant evidence of compositional or functional change, while there was a significant increase in structural diversity.

INTRODUCTION

Habitat is the portion of the landscape that contains the materials organisms need to survive (Garshelis, 2000) and is delineated at a variety of hierarchical, spatial, and temporal scales (Frissell et al., 1986). The spatial scale is tied inherently to major drivers and temporal periodicities of change, as turnover is fastest on the smallest scale and slowest at the largest scale. Meanwhile, ecosystem functions also are inherently linked to habitat and are driven by similar temporal and spatial levels. Nutrient cycling takes place over reach scale or larger (Newbold et al., 1982), whereas stream re-aeration takes place at the macrohabitat or lower (Bales and Nardi, 2007). Habitat scale has been linked to community composition as well; macroinvertebrate assemblages are the result of individual species from the regional pool, which have passed through scale-dependent filters that restrict others from persisting at a given site (Poff, 1997).

Despite habitat's relevance, measurements of habitat are neglected in disturbance ecology as the response focus is primarily on population and community level impacts and the resulting biomass (Hughes et al., 2007). However, some studies have sought to address the indirect effects disturbance can have through habitat destruction and degradation. For example, Templeton et al. (2011) examined the impact of forest fires on lizard distribution, finding increased dispersal between open glade patches coincident with prescribed burns in the surrounding forest matrix. As such, we can approach disturbance and recovery in a

holistic sense, where habitat recovery would follow similar trends to the biotic response – namely driven by disturbance type, intensity, frequency, and area (Nimmo et al., 2015).

Habitat recovery as part of restoration activity becomes especially important in the context of alternate stable states. For example, if restorative actions are a pulse type disturbance, then the creation of suitable habitat should arise after a brief disruption period rather than a recovery of the pre-existing landscape. As opposed to typical pulse disturbances where the recovered state is similar in composition to the previous state, restoration projects aim to force the system into an alternate compositionally distinct state (Wohl et al., 2005). Thus, a successful restoration should see the development of suitable habitat from the initial degraded state with an intervening reorganization period. This is perhaps the mechanism by which biotic response typically proceeds under the “Field of Dreams Hypothesis” (Sudduth et al., 2011) namely “if you build it they will come,” which itself is built on Poff’s (1997) prediction of environmental filtering. Even so, there is contradictory data about whether simple habitat availability is enough to allow recolonization by lost taxa (Sudduth et al., 2011).

Because the definition of habitat is for specific species or specific guilds (Hall et al., 1997), a working definition of community level habitat is hard to establish. The difficulties include limited information on typical and specific use of habitat (Krausman, 1999). This challenge is why many choose to consider habitat as a vegetative class, above the patch scale and below the landscape scale (Wang et al., 2014). Using the biodiversity framework set forth by Noss (1990), measures of habitat – however defined – can be broken down into one of three attributes: compositional, functional, and structural. These three components of habitat provide animal assemblages with the template for survival within

an ecosystem: patch presence, functional requirements, and patch distribution respectively. In addition, this categorization leaves open the possibility of assessing each attribute on multiple spatial scales. While all three components address different aspects of habitat benefit to community assemblages, a combined metric of habitats' composition, structure, and function can be calculated to assess the relative heterogeneity of the system (Dimino et al., *in prep*).

As previously highlighted, it is critical to consider the effect of scale on habitat features relevant to the assemblage being investigated (Death and Joy, 2004). Therefore, an accurate assessment of habitat changes from restoration requires a variety of scales to be used, such as the watershed (i.e. landscape) and reach (i.e. local) scale. At the landscape scale, habitat composition can be approximated as land use/land cover percentages within the watershed. Structurally, the fragmentation of the natural space patches or the corridor networks and the degree of network branching within the river system provide information on watershed level patterning of available habitat. Meanwhile, impervious cover percentages provide an integrated measure of functional barriers to colonization, due to the cumulative effects of temperature, organic pollution, and increased flashiness of rivers in highly impervious areas. At the reach (local) scale, composition can be measured through the abundance and diversity of in-stream cover types and river bottom substrates. River sinuosity within a system or a reach acts as a structural assessment of spatial variation. Discrete units of sub-reach scale habitat, known as macrohabitats, contain their own subset of interlinked functional processes, including decomposition within pools and oxygenation at riffle beds (Frissell et al., 1986; Poff, 1997). These reach scale habitat measures can be combined into a holistic assessment of in-stream habitat heterogeneity (Dimino et al., *in*

prep), but also may be investigated independently as areas of potential habitat improvement in one component (*sensu* Noss, 1990) of biodiversity.

STUDY AIM AND HYPOTHESES

Site-wide restoration involves more than simply the stream channel and thus inherently affects processes at both a local and sub-watershed scale. Therefore, the watershed scale impact of riparian restoration and the transition of land use was unlikely to drive stream habitat changes as much as the local scale direct changes to channel geomorphology and introduction of large woody debris. However, the level to which the land use has changed the composition and structure of the sub-watershed land cover requires quantification in order to account for any contribution this may have had to observed changes.

The goal of this study was to determine if restoration resulted in any quantifiable changes to the habitat, at the landscape (*i.e.* watershed) and local (*i.e.* reach) scale, and how those changes compared to control sites of high and low habitat diversity. To achieve this goal, I explored 2 primary hypotheses.

First, I expected that the local scale habitat between the treatments would be significantly different due to the direct effects of the adjacent land uses and history of channelization, sedimentation, and riparian zone modification associated with intensive agricultural use (Maloney and Weller, 2011).

Second, I predicted that over the course of the 3-year study, changes to the local scale habitat would occur, specifically an increase in both habitat composition and structure.

However, I expected that an increase in functional habitat would be unlikely, as several studies have noted the difficulty in altering functional aspects of ecosystems (Alsterberg et al., 2017; Palmer et al., 2014) and other restorations have had problems in ameliorating lost habitat features, such as reduced embeddedness and undercut bank stability (Shields et al., 2003).

To establish a baseline for watershed changes, I computed before-after comparisons for watershed characteristics. In order to test the local scale predictions, I sampled reach-based parameters at three treatments across three years and analyzed temporal changes through effect size analysis, while comparing between treatments via PCA and ANOVA. My findings suggest that watershed scale land use composition and fragmentation exhibited larger than expected potential for increase, while at the local scale observed heterogeneity changes were limited to increased depth variability and decreased substrate variability.

MATERIALS AND METHODS

Study Area

Research was conducted at three (3) treatments, representing directed pulse disturbance (Cranberry Bog Restoration, Tidmarsh Farms, CBR), background disturbance (Least impacted reference, Mashpee River, LIR), and an agricultural press disturbance (Active Cranberry Bog, ACB) in three separate low-order (1st-3rd) streams fed by primarily groundwater and drainage fed ponds in the Atlantic Coastal Pine Barrens Ecoregion of Massachusetts (Figure 1). The watershed composition of land use/land cover and

fragmentation for each treatment were compiled (Table 1). CBR underwent active restoration in Winter 2015/2016. Each treatment consists of three (3) replicate stations defined as a 100m length segments where data was collected. Each station was separated by at least 100m in river channel length.

Watershed Characterization

Treatment watersheds were characterized using data from prior to 2015, allowing for comparisons of treatment watersheds pre-restoration. Total watershed delineations were determined for the treatment level drainages (identified as the downstream end of the furthest downstream station) using USGS's StreamStats service (United States Geological Survey, 2016). These StreamStats watersheds were imported into ArcGIS (ESRI) for further analysis. Additional land use, impervious surface coverage, and elevation maps were obtained through MassGIS (Bureau of Geographic Information, 2016). All data obtained reflected landscape level values predating the current study.

Calculation of watershed parameters was done in ArcGIS (ESRI) by clipping obtained rasters onto the treatment watershed and calculating values for each treatment. Total drainage area, land use percentages, average impervious cover percentage, and average elevations were obtained for each treatment to establish baseline features of all treatments for future comparisons. Forested and natural land use was analyzed for fragmentation, calculated from land use rasters using the *raster* package (Hijmans, 2018; R Core Team, 2018) including a combined metric of total area, edge composition, patch number, and patch isolation (Bogaert et al., 2000). These same values were recalculated for CBR once the total restoration area was reclassified from agricultural land to natural use.

Local Scale Habitat Assessment

Habitat assessments were conducted in order to quantify between treatment differences at the local scale. A Basin Area Stream Survey (BASS) habitat inventory was conducted seasonally at all 100m reach stations over the 3 year study (McCain et al., 1990). Briefly, BASS employs delineating macrohabitats (i.e. riffles, runs, and pools) and performing cross-stream transects of each available macrohabitat. BASS measures include number of macrohabitats, length of each macrohabitat, width and depth profile of each habitat, as well as estimation of in-stream habitat structures and bank parameters. Values were recorded in the field and analyzed by principle component analysis using the *vegan* package in R (Oksanen et al., 2018; R Core Team, 2018) to determine variability between treatments. Station groupings were tested through both guided and unguided clustering to evaluate *a priori* treatment assignments to determine if treatment group explains observed patterning.

The raw BASS data also was used to generate composite measures of in-stream habitat diversity – Shannon-Weiner diversity of substrate, Shannon-Weiner diversity of in-stream cover, number of macrohabitats per 100m, and reach standard deviation of depth (Dimino et al., *in prep*). Each parameter of the metric represents an attribute of biodiversity (e.g. composition, structure, or function), allowing for determination of not only overall habitat shifts, but also the local change drivers that may be acting within a system.

These individual parameters were analyzed for effect size of change relative to control treatment following effect size for a BACI design:

$$d = (A_I - A_C) - (B_I - B_C)$$

Where A represents the mean parameter value in the After period while B represents the mean value for the parameter Before (e.g. in 2015, the initial year of study) for both I , impact treatment, and C , control treatment (Christie et al., 2019). Confidence intervals (95%) were generated based on these effect sizes and pooled variance:

$$s_p^2 = \frac{(n_{BI} - 1) * s_{BI}^2 + (n_{BC} - 1) * s_{BC}^2 + (n_{AI} - 1) * s_{AI}^2 + (n_{AC} - 1) * s_{AC}^2}{(n_{BI} + n_{BC} + n_{AI} + n_{AC} - 4)}$$

$$CI = d \pm 1.96 * \sqrt{\frac{s_p^2}{n_{BI}} + \frac{s_p^2}{n_{BC}} + \frac{s_p^2}{n_{AI}} + \frac{s_p^2}{n_{AC}}}$$

Where s^2 and n represent the variances and sample size for each set of treatment/period combinations (Christie et al., 2019). This calculation was done for both 2016 and 2017 as separate After periods, as there is no assumption of a stepwise change immediately following restoration combination, comparable to a Before-During-After-Control-Impact study (Roedenbeck et al., 2007). Additionally, due to the differences between control sites, separate analyses were run treating ACB or LIR as the control site. These analyses were completed for all 4 parameters. Effect sizes were determined to be statistically significant if the confidence interval did not overlap zero.

In order to assess changes at CBR relative to LIR and ACB conditions, Welsh's t -tests were computed pairwise within years, comparing CBR to ACB and CBR to LIR, to determine if CBR's state altered relative to either reference condition. To increase power

without increasing Type I error, pairwise comparisons between LIR and ACB were not conducted. Significance was adjusted due to multiple comparisons using Bonferroni's correction ($\alpha=0.008$, $n=6$) (Legendre and Legendre, 2012).

In addition to field-collected station data, ArcGIS was used to calculate stream sinuosity from aerial photography both before and after the restoration event at all stations. Sinuosity was calculated as the actual distance in river divided by the straight-line distance, as determined through upstream and downstream station GPS coordinates. Measured sinuosity was then analyzed for differences using one-way ANOVA, with treatment (ACB, LIR, and CBR before and after restoration) acting as predictors ($n = 3$ per treatment).

RESULTS

Watershed Characterization

Computation of watershed scale parameters provided important baseline data. In particular, treatment watersheds ranged from 7.66km² to 42.25km² in total drainage, have between 50.26 and 75.03% Forested/Natural land use, and range between 1.98 to 9.95% watershed-wide average impervious cover (Table 1). Fragmentation, measured as a combination of land use percentage, proportion of edge habitat, number and isolation of patches, was between 136.57 and 166.73 on a scale of 0 to 200 (Table 1).

When reclassifying the CBR site from cranberry agriculture to Forested/Natural after the 2015 restoration (Figure 2), the changes to the watershed level land use are evident (Table 2). Briefly, total Forested/Unimpacted land use increases by nearly 10 percentage points, with an identical drop in cranberry agriculture. Fragmentation also decreases,

largely driven by the increase in Forest/Natural land use and secondarily by the decrease in edge habitat proportion (Table 2).

Local Scale Habitat Assessment

Habitat Characterization PCA. Local scale habitat measures were expected to demonstrate separation of all three treatments following ordination, as well as a separation of post-restoration CBR from pre-restoration clustering. Principal component analysis of local scale habitat variables demonstrate a number of findings, including that habitat differentiation is robust to seasonal differences, there is separation of LIR from CBR and ACB occurs along PC1, determined via ANOVA, and that ACB habitat data groups within the variation shown at CBR.

All stations were analyzed together and by treatment to determine if seasonality affected the results. First, even though sampling was conducted seasonally, there was no clear patterning of the data, either in bulk (Figure 3) or by treatment (not shown) to suggest seasonality plays a critical role in the differentiation of the treatments by habitat. As a result, subsequent analysis of the habitat surveys via PCA did not separate sites out based on season of collection (Figure 4). Secondly, LIR treatment stations show separation from the CBR and ACB stations along PC1 as determined by ANOVA ($F=560$, $df=2$, $p<0.001$) and subsequent post-hoc Tukey's HSD ($p<0.001$). PC1 is correlated negatively with habitat length and percent clinging vegetation, while PC2 is moderately correlated with habitat length, wetted width, and clinging and rooted vegetation (Table 3). Thirdly, all ACB sites appear as a subset of the variation exhibited by the CBR sites both before and after restoration. Finally, we observed an increase in variance from the pre-restoration state in 2016, followed by a return in 2017 to values more similar to the pre-restoration state.

Habitat Heterogeneity. Local scale measures of compositional and structural heterogeneity were expected to increase following restoration while functional change was not expected to occur within the timespan of the study. When analyzing the Habitat Heterogeneity parameters, only standard deviation of depth in 2017 demonstrated an effect size confidence interval that did not overlap zero. relative to both pre-restoration and in comparison, to reference conditions. When comparing CBR to the reference conditions, changes in both substrate diversity and diversity of depth were observed.

In-stream cover diversity increases following restoration and continues to increase over the course of the study ($d_{LIR} = 0.119$, CI: [-0.297,0.534]; $d_{ACB} = 0.067$, CI: [-0.294,0.428]) in 2016, ($d_{LIR} = 0.290$, CI: [-0.210,0.788]; $d_{ACB} = 0.153$, CI: [-0.294,0.600]) in 2017, but is not statistically significant (Figure 5A). Pair-wise Welch's corrected t-tests within year show no significant differences between CBR and the other two treatments in any year (Welsh's t-tests, Appendix A.2) (Figure 5B). Substrate compositional diversity falls immediately following restoration in 2016, resulting in a statistically significant effect size change relative to LIR, but not ACB ($d_{LIR} = -0.387$, CI: [-0.750, -0.024]; $d_{ACB} = -0.240$, CI: [-0.533,0.053]). However, in 2017 the substrate composition increased and was no longer significantly different relative to either control ($d_{LIR} = -0.275$ CI: [-0.629,0.079]; $d_{ACB} = -0.059$, CI: [-0.419,0.301]) (Figure 6A). CBR is not significantly different from either other treatment in 2015 or 2017, but is significantly different than LIR in 2016 (Welsh's $t = -5.295$, $v = 3.874$, $p = 0.007$) (Figure 6B). Depth standard deviation significantly increases following restoration in 2016 ($d_{LIR} = 0.174$ CI: [0.064,0.285]; $d_{ACB} = 0.192$, CI: [0.065,0.319]) and 2017 remains significantly different relative to ACB control but not LIR ($d_{LIR} = 0.084$ CI: [-0.001,0.169]; $d_{ACB} = 0.152$, CI: [0.097,0.206]) (Figure 7A). Pair-wise

t-tests show no significant differences between CBR and LIR or ACB for any years (Welsh's t-tests, Appendix A.2) (Figure 7B). Finally, the number of macrohabitats per 100m at CBR increases in the first year post restoration ($d_{LIR} = 0.517$ CI: [-1.798,2.833]; $d_{ACB} = 1.352$, CI: [-0.271,2.976]), followed by a slight decline in 2017 ($d_{LIR} = -0.494$ CI: [-2.331,1.342]; $d_{ACB} = 0.622$, CI: [-0.058,1.303]), but neither are significant (Figure 8A). Pair-wise t-tests demonstrate CBR as significantly different than LIR in 2015 (Welsh's $t = -8.014$, $v = 3.000$, $p = 0.004$) and 2017 (Welsh's $t = -7.475$, $v = 3.329$, $p = 0.003$) (Figure 8B).

Sinuosity. Station sinuosity at CBR was expected to increase as a result of the rechannelization process. Channel morphology changes after restoration did demonstrate an increase in sinuosity post-restoration, however this difference is not statistically significant ($F = 2.832$, $df = 3$, $p = 0.10$) (Figure 9).

DISCUSSION

Overall, the results of my study lead to three main findings. First, restoration of the former cranberry bog at CBR resulted in a net increase in habitat potential as measured by land use change and increased habitat connectivity, greater than our initial expectation. Second, for the local scale habitat characterization PCA, the LIR sites showed clear separation from either ACB or CBR, both before and after the restoration, partially confirming our hypothesis of local scale habitat differences. CBR did increase in local scale habitat heterogeneity following restoration, but without substantially changing its similarity to pre-restoration channel morphology, and neither before nor after restoration CBR values were distinct from ACB. Finally, when habitat heterogeneity metrics were

investigated, local scale changes did occur, but were limited to declines in substrate diversity and an increase in depth variation, both properties that are features of ACB rather than LIR.

Watershed Characterization

Although fragmentation overall is variable between the 3 systems, there is no clear evidence in either land use proportions or impervious cover to suggest that CBR exists in a particularly degraded watershed in comparison to the other two treatments. Indeed, from a watershed scale perspective, ACB exhibits parameters that, counter-intuitively, may suggest the highest level of water quality and taxonomic diversity. Watershed level parameters, however, only partially filter the regional species pool, while local effects play a more selective role (Poff, 1997).

Land use reclassification shows that CBR's riparian zone restoration could result in an increase in forested/natural land and decrease in fragmentation by nearly 10% and 5% respectively. However, it is important to note that these changes represent the maximal watershed level changes the restoration could make, following total reforestation and wetland development and the removal of all manmade barrier structures on site at CBR. Indeed, using watershed forest cover as a proxy for stream habitat connectivity may account for limitations due to adult dispersal of some insect species, but has limited utility for fish and non-flighted freshwater macroinvertebrates (Smith et al., 2015). Additionally, even maximal change in the watershed with respect to forest cover and connectivity does not reduce urban influence and impervious cover within the watershed, a noted driver of in-stream degradation (Morse et al., 2003).

Local Scale Habitat Assessment

The BASS habitat characterization PCA of transect data from all three treatments shows the expected separation between the unimpacted reference (LIR), and both the active and restored cranberry bog. There was no clear separation between CBR and ACB within the PCA, contrary to our expectation that CBR would show unique habitat befitting its 5 years of passive restoration. PCA separation was driven largely by the relative length of the macrohabitats and the percentage of clinging vegetation. Both variables are tied to the anthropogenic impact cranberry farming imposes upon the system, through channelization and loss of riparian canopy cover, respectively (Kristensen et al., 2013; Sudduth et al., 2011). This may demonstrate the legacy effects of agriculture (Maloney and Weller, 2011) and the inability of low gradient streams to adequately flush sediments and reshape stream channels (Waite, 2014). Much of the variability in CBR sites was found in the first year post-restoration, consistent with both a disturbance event and potential reorganization period (Stanley et al., 2010). Although there was no distinct separation of habitat, variation along PC2 does demonstrate some differences between ACB and CBR, especially for the first year of restoration. The associated variables of habitat length, channel width, and in-stream vegetation can be tied to the direct manipulation of the channel during the restoration.

When examined for individual contributions to ecosystem level biodiversity, the components of the Habitat Heterogeneity model provide a complex picture. In-stream cover and substrate diversity can be considered aspects of compositional diversity due to the niches they create (Bond and Lake, 2005). While neither in-stream cover nor substrate diversity show statistically significant shifts following restoration, both change in

important ways. In-stream cover monotonically increases into the second year, allowing the possibility of sustained change in years following the study. Substrate diversity, on the other hand, declines in the first year and rebounds in the second, potentially operating as a pulse response to the rechannelization. This lack of change in compositional diversity runs contrary to my prediction of increased habitat compositional diversity as a result of the restorative actions, although significant changes may be masked by small sample size and high variability. Other studies that have found increases in composition post-restoration have either assessed change later following restoration (Friberg et al., 1998; Kupilas et al., 2017) or monitored specific features following targeted interventions (e.g. width and flow changes following woody debris removal, Dumke et al., 2010). As a measure of structural diversity, standard deviation in depth was expected to be highest for the LIR treatment, as a diversity of depth may support a more diverse fish and invertebrate community (Lake, 2003). However, the extreme variation in depth, from near the surface along the bank to over a meter depth at the thalweg makes the channels at CBR post-restoration and ACB sites exhibit a more diverse depth profile than LIR. Post-restoration, CBR's depths became more varied, as the channel was flattened at the banks to promote flood plain interaction while deepening at the thalweg to interface with the peat layer below the former farm surface (A. Hackman, personal communication). Thus, CBR did exhibit structural increases in depth profile, but it remains to be seen whether this change will persist in long term channel profiling. Finally, number of macrohabitats acted as a proxy for functional change, as macrohabitats have distinct microhabitats and alterations to water chemistry that may impact aquatic organisms (Frissell et al., 1986). Although macrohabitats increased at CBR following restoration, the increase was not able to be deemed statistically significant.

Thus, my hypothesis that CBR would experience no functional change over the course of the study was upheld, although the increase observed in the first 2 years may bear out as sustained and quantifiable functional change with continued monitoring.

As a final measure of structural change to the system, the post-restoration sinuosity increase was found to be not significantly different. Sinuosity has many functions within the stream channel, including decreasing sedimentation, slowing overall flow, and creating specialized pool habitat (Brookes, 1987). The lack of power due to limited sample size may explain why the increase was not detectable.

The changes observed in the habitat suggest the restoration may have created initial changes in compositional, structural, and functional diversity, beyond what I predicted as initial change only in the compositional attribute of habitat diversity. Given the lack of statistical power to determine significant shifts in many cases, whether the functional change persists after the reorganizational period remains to be seen. Many studies have suggested that functional change is necessary to accommodate community shifts, while compositional changes are more likely to occur first (Palmer et al., 2014). Ecosystem function is also highly tied to ecosystem resistance and resilience, potentially allowing community shifts to follow.

CONCLUSIONS

As restorations become more common tools to restore ecosystem functioning and promote biodiversity, the change in habitat at multiple scales cannot be ignored. Stream and terrestrial habitat operate at both the local scale and at the watershed scale, effecting

key features such as population persistence or insect dispersal respectively. As such, restorations that affect a significant percentage of the watershed must be investigated for both their local and regional impact. The alteration of nearly 10% increase in natural categories of land use and habitat connectivity is not insignificant, although it should be noted that this represents the upper bound for the landscape level effects of the restoration – namely the immediate incorporation of the total land area of CBR into a broadly classed level of natural habitat. Nevertheless, this change in habitat at the watershed scale is important to keep in mind when examining local shifts in habitat or community dynamics as well. At the reach scale, the characteristics of transect habitat suggest that restoration did not significantly alter the legacy of farming within the span of the 2-year study. When individual components of reach scale habitat were investigated, there are mixed effects, including no change in macrohabitat diversity or in-stream cover diversity, transitory change in substrate diversity, and only depth variability remaining changed after 2 years, having increased in similarity to ACB in this regard. Notably however, we do see the highest changes in the first year of the restoration, consistent with changes expected from a pulse type disturbance. The high variability in habitat parameters at CBR in both years post-restoration also suggest that the system has not yet stabilized (Carpenter and Brock, 2006).

Although 2 years following restoration is a relatively short study and does not account for long-term changes to the riparian zone through reforestation, the activities undertaken as part of the active restoration were expected to present immediate and persistent changes to the reach scale habitat, through increases in sinuosity and the incorporation of woody debris. The effect however appears to demonstrate a system still

in flux regarding habitat parameters and some evidence of pulse disturbance rather than maintained change.

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TABLES

Table 1: Watershed characterization for each treatment, obtained from spatial data before 2015. ACB – Active Cranberry Bog, CBR – Cranberry Bog Restoration, LIR – Least Impacted Reference Condition.

Treatment	ACB	CBR	LIR
Area (km ²)	18.23	7.66	42.25
<u>Land Use/Land Cover</u>			
% Forest/Natural	75.03	50.26	55.57
% Non-Cranberry Agriculture	0.18	0.24	0.60
% Cranberry Agriculture	1.88	10.45	0.13
%Urban	2.68	23.28	10.59
% Impervious Cover	1.98	9.95	6.30
Fragmentation (ϕ) ¹	166.73	136.57	143.29
<i>Proportion of Historic Habitat (α)</i>	75.03	50.26	55.57
<i>Boundary Length (β)</i>	96.00	75.37	89.64
<i>Number of Patches (ν)</i>	99.95	95.55	92.79
<i>Patch Isolation (δ)</i>	54.42	36.22	28.30

¹ Fragmentation metric from Bogaert et al. (2000) for forest/natural land use. Fragmentation is a combined, normalized metric on a scale from 0 to 200, representing maximum and minimum fragmentation respectively. The 4 components of this metric, range from 0 to 100 representing maximum to minimum fragmentation respectively.

Table 2: Changes to watershed parameters when the restoration site (CBR) is recoded from Cranberry Farm land use to Forested/Natural land use.

Treatment	CBR Pre-Restoration	CBR Post-Restoration	Change ¹
<u>Land Use/Land Cover</u>			
% Forest/Natural	50.26	60.12	9.86
% Non-Cranberry Agriculture	0.24	0.24	0.00
% Cranberry Agriculture	10.45	0.66	-9.79
%Urban	23.28	23.28	0.00
Fragmentation (ϕ) ²	136.57	144.91	8.34
<i>Proportion of Historic Habitat (α)</i>	50.26	60.12	9.86
<i>Boundary Length (β)</i>	75.37	80.81	5.44
<i>Number of Patches (ν)</i>	95.55	97.08	1.53
<i>Patch Isolation (δ)</i>	36.22	37.81	1.59

¹ Total change in land cover does not sum to zero due to slight differences in raster creation between raw land use maps obtained from MassGIS and the rasterization of land parcel data used to delineate CBR's boundaries.

² Fragmentation metric from Bogaert et al. (2000) for forest/natural land use. Fragmentation is a combined, normalized metric on a scale from 0 to 200, representing maximum and minimum fragmentation respectively. The 4 components of this metric, range from 0 to 100 representing maximum to minimum fragmentation respectively.

Table 3: Loading values for each original variable on the first two principle components after PCA analysis of the local scale habitat data from all three treatments across all three years.

Variable	PC1 (33.04%)	PC2 (10.03%)
Length of Habitat	-0.7813222	0.56131155
Wetted Width	0.18867467	0.54345261
Thalweg Depth	-0.4353481	-0.303593
Minimum Depth	-0.3508264	0.03180419
Standard Deviation of Depth	-0.3143273	-0.3162003
% Boulder Substrate	0.03491382	0.10338119
% Cobble Substrate	0.0179058	-0.1769267
% Gravel Substrate	0.3209406	0.09894388
% Sand Substrate	-0.1085014	-0.0468564
% Fine Sediment Substrate	-0.261565	-0.0232071
% Under Cut Bank	0.1614075	-0.1441662
% Large Woody Debris	0.00951469	-0.3575576
% Small Woody Debris	0.13093508	-0.2112439
% In-stream Terrestrial Vegetation	0.08023346	-0.0702898
% Clinging Vegetation	-0.7813222	0.56131155
% Rooted Vegetation	0.18867467	0.54345261

FIGURES



Figure 1: Experimental treatment study locations and their relative locations in Southeastern Massachusetts. From bottom to top on right: Tidmarsh Farms (CBR, pre-restoration imagery), an active flow-through cranberry bog (ACB), and the Mashpee River flowing through the Trustees of Reservations conservation land (LIR) (ESRI). Map made by S.McCanty.

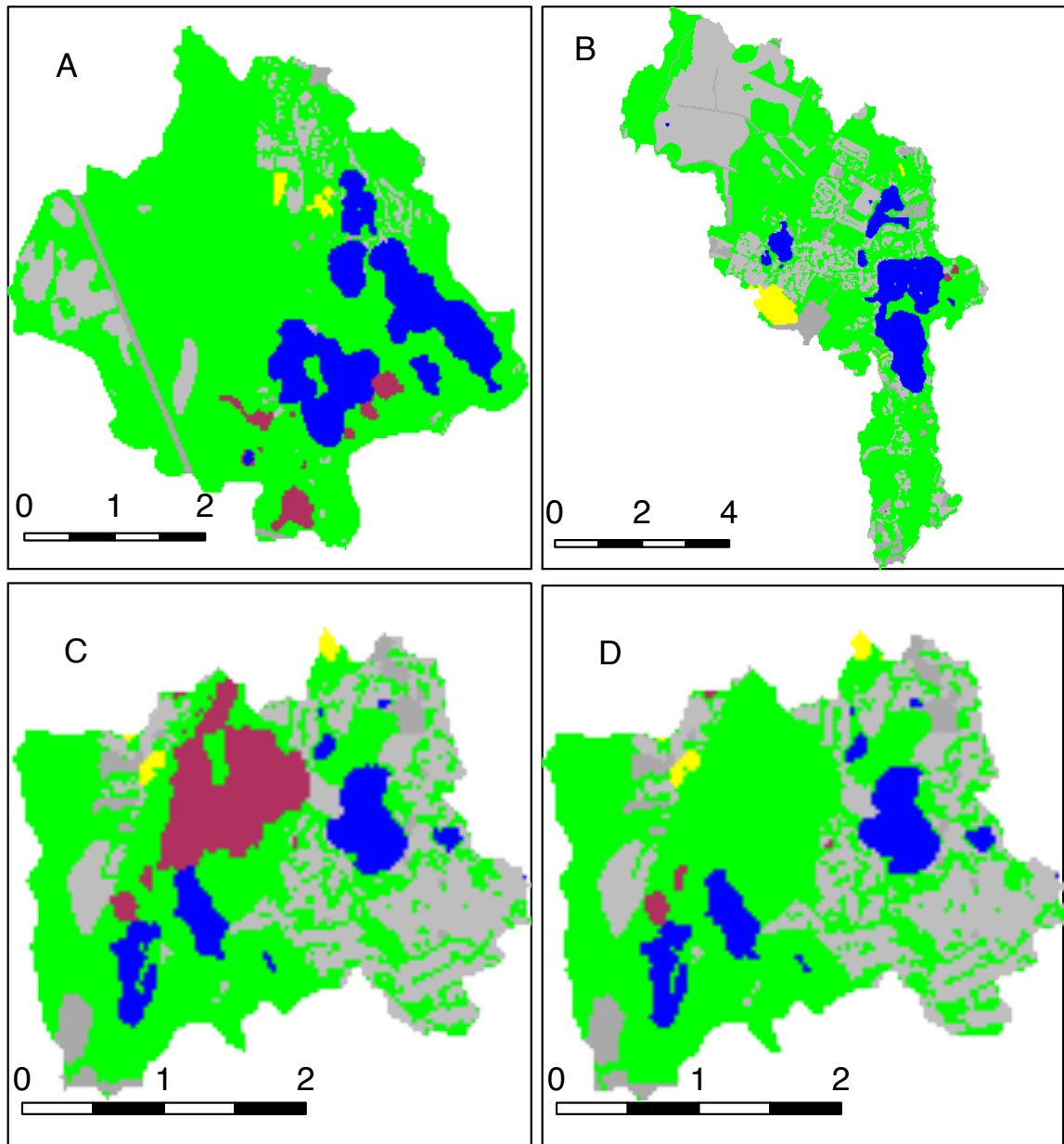


Figure 2: Land use fragmentation map for ACB (A), LIR (B), and CBR pre- (C) and post-restoration (D). The change from C to D is driven by the recoding of the restoration area from Cranberry agriculture to Forested/Natural. Scale bars represent distance in kilometers. Color scheme: Green – Forested/Natural land use, Blue – Water, Gray – Urban/Industrial land use, Yellow – Agricultural land use, Maroon – Cranberry land use. Land use data obtained from MassGIS (Bureau of Geographic Information, 2016) and visualized in R (Hijmans, 2018; R Core Team, 2018). Map made by S.McCanty.

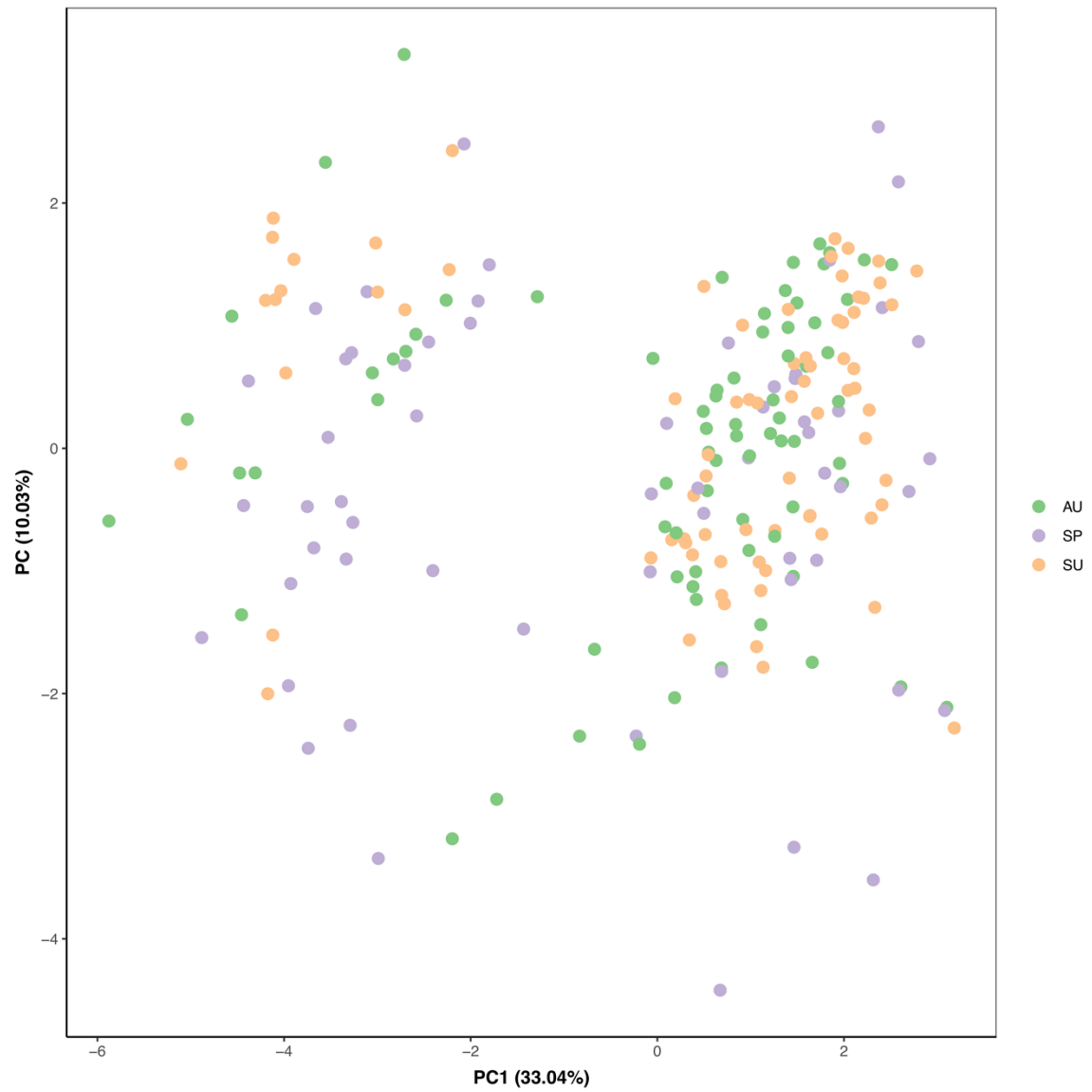


Figure 3: PCA of local habitat transect data (BASS) from 2015-2017, color coded by samplings in spring (green), summer (blue), and autumn (red).

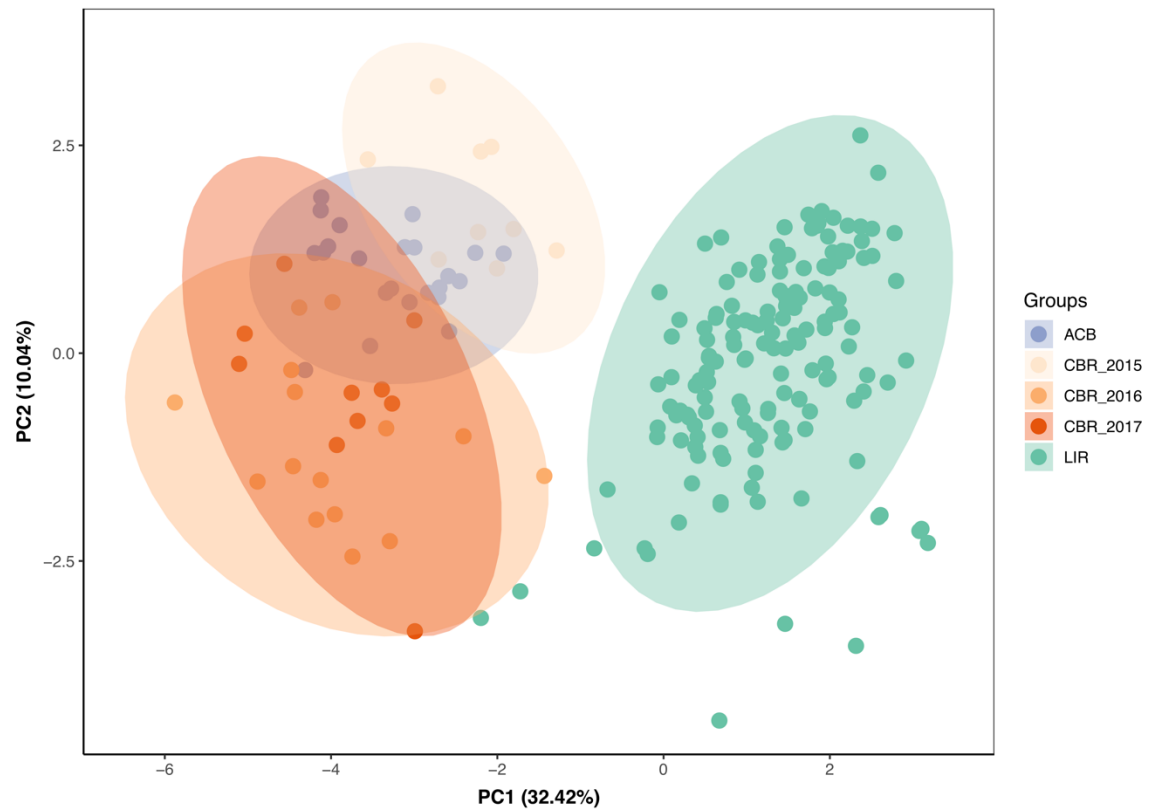


Figure 4: PCA of local habitat transect data (BASS) from 2015-2017, color coded by treatment, with 95% confidence ellipses for group identification.

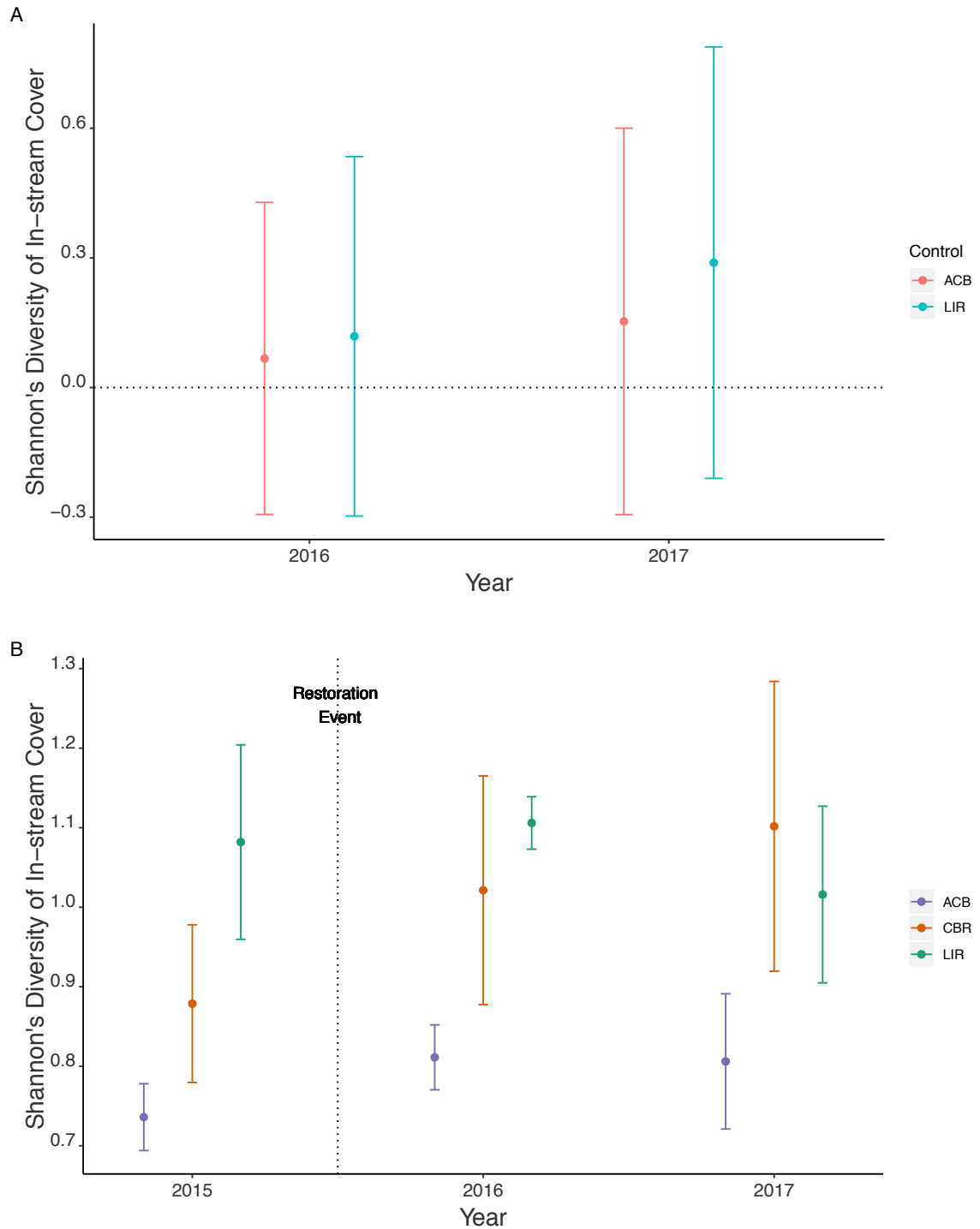


Figure 5: Shannon's Diversity of "In-stream Cover" as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests ($\alpha = 0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

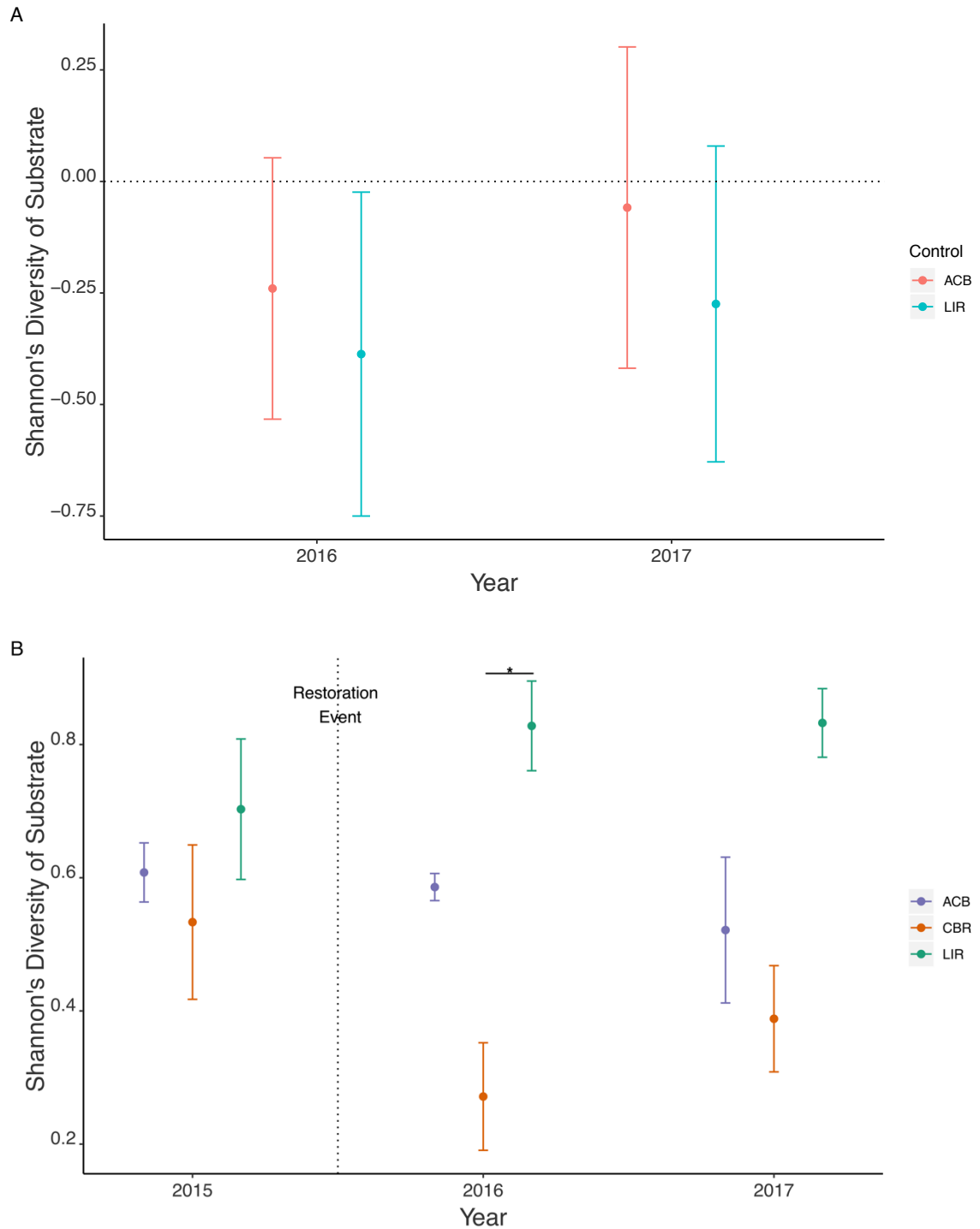


Figure 6: Shannon's Diversity of "Substrate" as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests ($\alpha=0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

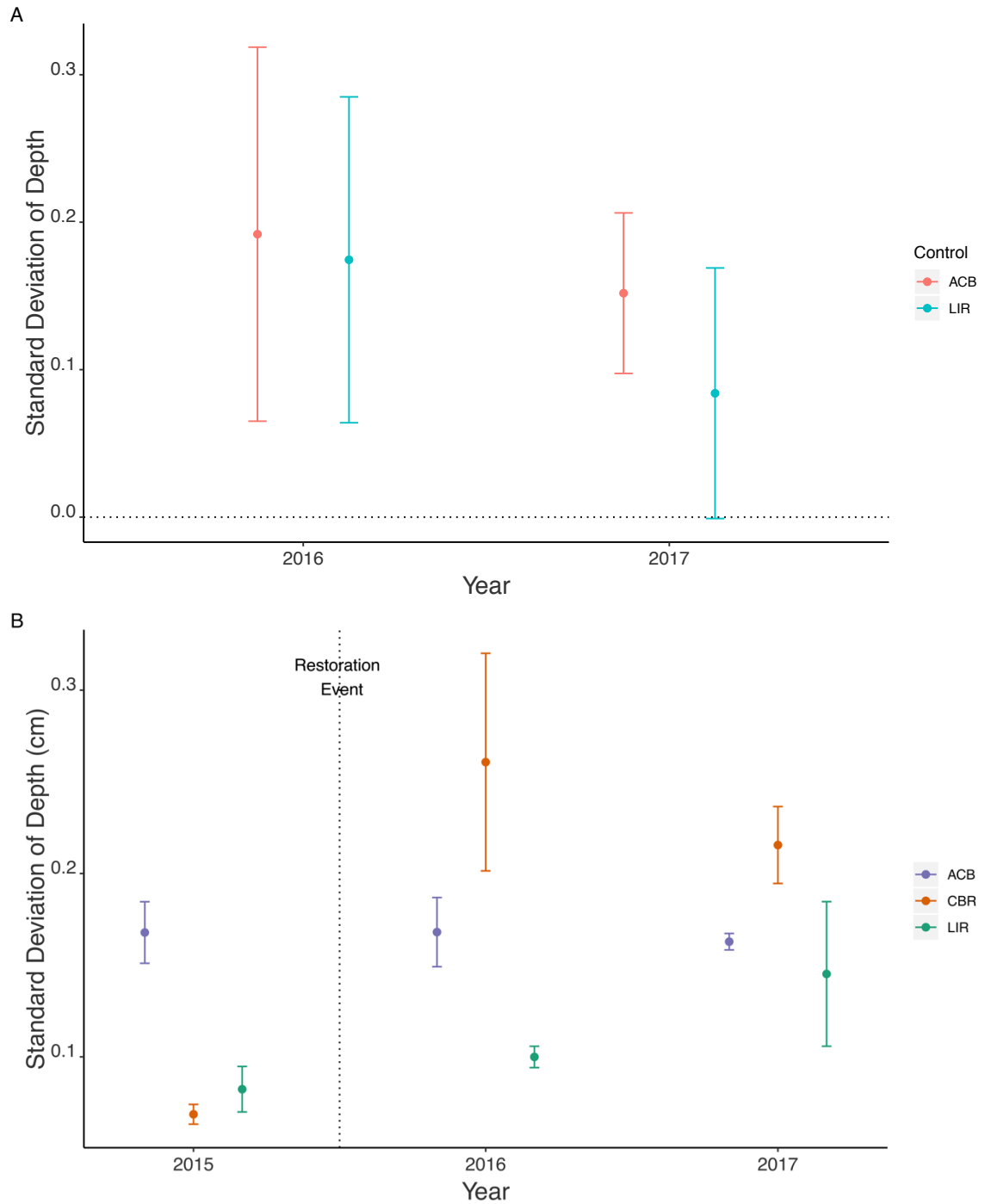


Figure 7: Standard Deviation of “Depth” as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch’s t-tests ($\alpha=0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

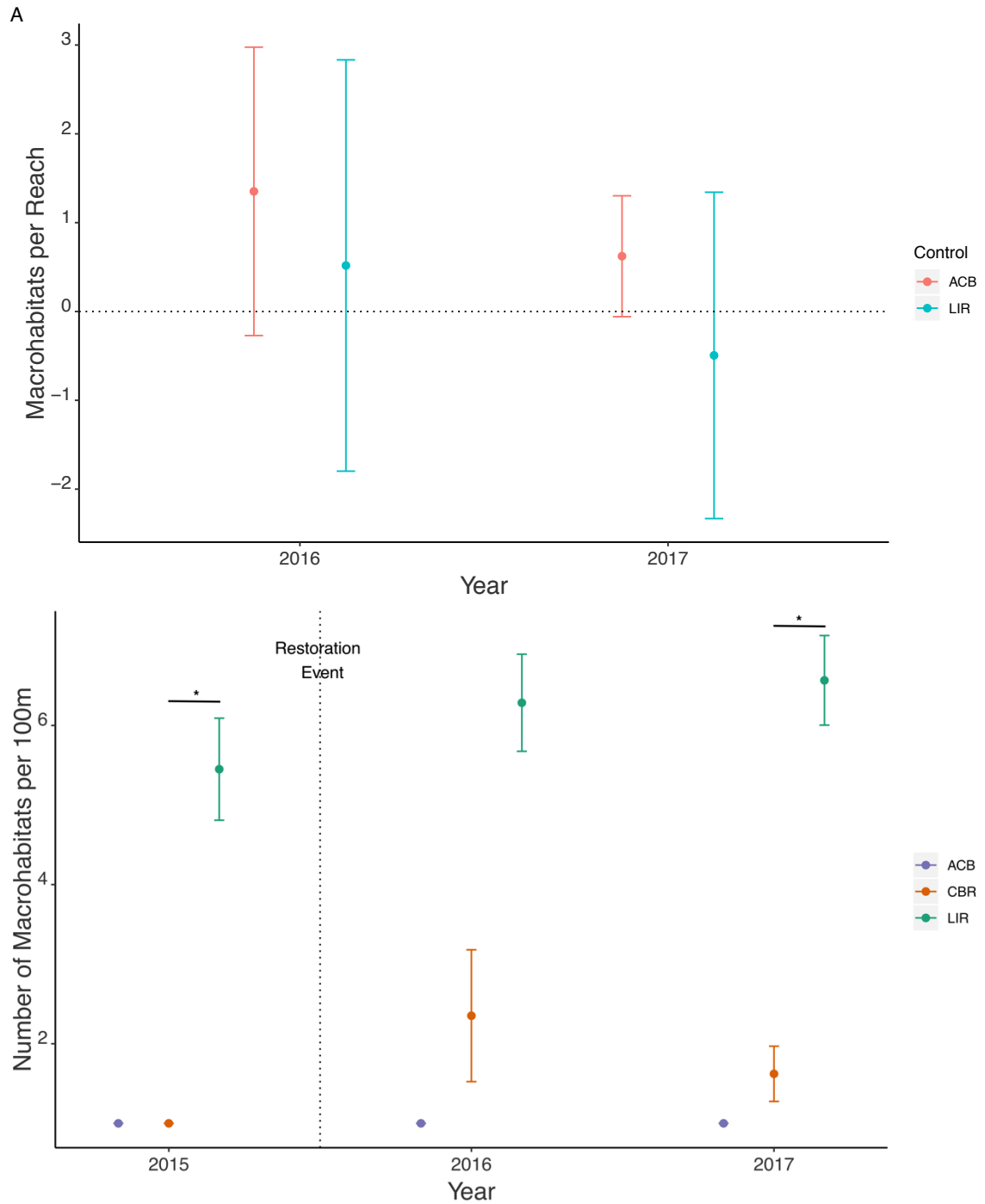


Figure 8: Number of macrohabitats per 100m as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests ($\alpha = 0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

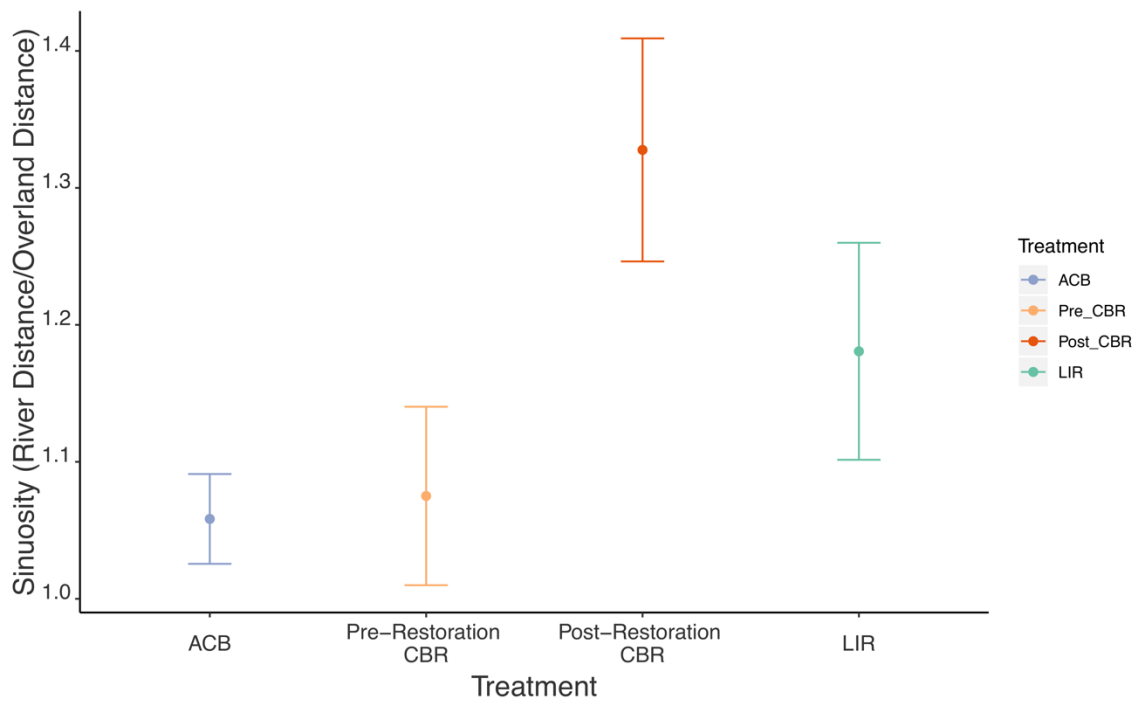


Figure 9: Sinuosity with standard error at ACB (n=3) and LIR (n=4), as well as pre- (n=3) and post-restoration (n=3) at CBR. Treatments were not significantly different. ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

APPENDIX A

Table A.1: Effect size (BACI) of change before and after restoration (2015 to 2016, or 2015 to 2017) relative to control (LIR or ACB) treatment. Each effect size is calculated as described in-text, with 95% confidence intervals generated from pooled variance. Bolding represents effects with confidence intervals not overlapping 0. ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

Year	Control	<u>H' In-stream Cover</u>		<u>H' Substrate</u>	
		d	95% CI	d	95% CI
2016	LIR	0.119	[-0.297,0.534]	-0.387	[-0.750,-0.024]
2016	ACB	0.067	[-0.293,0.428]	-0.240	[-0.533,0.053]
2017	LIR	0.289	[-0.210,0.788]	-0.275	[-0.629,0.079]
2017	ACB	0.153	[-0.294,0.600]	-0.059	[-0.419,0.301]

Table A.1 continued

Year	Control	<u>St. Dev of Depth</u>		<u>Number of Habitats</u>	
		d	95% CI	d	95% CI
2016	LIR	0.174	[0.064,0.285]	0.518	[-1.798,2.833]
2016	ACB	0.192	[0.065,0.319]	1.352	[-0.271,2.976]
2017	LIR	0.084	[-0.001,0.169]	-0.494	[-2.331,1.342]
2017	ACB	0.152	[0.097,0.0206]	0.622	[-0.058,1.303]

Table A.2: Within year differences between CBR and control treatments. T values represent Welch's two-tailed t-tests between CBR and listed treatment in the same year. Due to unequal variances, Welch's t-tests calculate ν , a modified df. Bolding represents changes that are significantly different ($\alpha = 0.008$, Bonferroni correction for 6 comparisons per year). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

Year	CBR:Treatment	<u>H' In-stream Cover</u>			<u>H' Substrate</u>		
		T value	P value	ν	T value	P value	ν
2015	ACB	1.325	0.286	2.696	-0.601	0.597	2.574
	LIR	-1.400	0.221	4.908	-1.149	0.312	4.181
2016	ACB	1.406	0.279	2.321	-3.778	0.053	2.253
	LIR	-0.574	0.619	2.210	-5.295	0.007	3.874
2017	ACB	1.471	0.243	2.831	-0.983	0.386	3.657
	LIR	0.402	0.712	3.307	-4.678	0.014	3.422

Table A.2 continued

Year	CBR:Treatment	<u>St. Dev of Depth</u>			<u>Number of Habitats</u>		
		T value	P value	ν	T value	P value	ν
2015	ACB	-5.625	0.019	2.410	-	-	-
	LIR	-1.136	0.315	4.294	-8.014	0.004	3.000
2016	ACB	1.489	0.255	2.400	1.633	0.244	2.000
	LIR	2.698	0.112	2.039	-3.823	0.022	3.676
2017	ACB	2.458	0.123	2.183	1.793	0.215	2.000
	LIR	1.574	0.212	3.048	-7.476	0.003	3.329

LITERATURE CITED

- Alsterberg, C., Roger, F., Sundbäck, K., Juhanson, J., Hulth, S., Hallin, S., and Gamfeldt, L. (2017). Habitat diversity and ecosystem multifunctionality—The importance of direct and indirect effects. *Sci. Adv.* 3, e1601475.
- Bales, J.D., and Nardi, M.R. (2007). Automated Routines for Calculating Whole-Stream Metabolism: Theoretical Background and User's Guide. USGS.
- Bogaert, J., Van Hecke, P., Eysenrode, D.S.-V., and Impens, I. (2000). Landscape Fragmentation Assessment Using a Single Measure. *Wildl. Soc. Bull.* 1973-2006 28, 875–881.
- Bond, N.R., and Lake, P.S. (2005). Ecological Restoration and Large-Scale Ecological Disturbance: The Effects of Drought on the Response by Fish to a Habitat Restoration Experiment. *Restor. Ecol.* 13, 39–48.
- Brookes, A. (1987). Restoring the sinuosity of artificially straightened stream channels. *Environ. Geol. Water Sci.* 10, 33–41.
- Bureau of Geographic Information (2016). Bureau of Geographic Information (MassGIS) (Commonwealth of Massachusetts, Executive Office of Technology and Security Services).
- Carpenter, S.R., and Brock, W.A. (2006). Rising variance: a leading indicator of ecological transition. *Ecol. Lett.* 9, 311–318.
- Christie, A.P., Amano, T., Martin, P.A., Shackelford, G.E., Simmons, B.I., and Sutherland, W.J. (2019). Simple study designs in ecology produce inaccurate estimates of biodiversity responses. *J. Appl. Ecol.* 56, 2742–2754.
- Death, R.G., and Joy, M.K. (2004). Invertebrate community structure in streams of the Manawatu–Wanganui region, New Zealand: the roles of catchment versus reach scale influences. *Freshw. Biol.* 49, 982–997.
- Dimino, T.F., McCanty, S.T., and Christian, A.D. (In Prep). Quantifying Habitat Heterogeneity in Low Gradient Headwater Streams using a Distance Based Model. *Diversity*.
- Dumke, J.D., Hrabik, T.R., Brady, V.J., Gran, K.B., Regal, R.R., and Seider, M.J. (2010). Channel Morphology Response to Selective Wood Removals in a Sand-Laden Wisconsin Trout Stream. *North Am. J. Fish. Manag.* 30, 776–790.
- ESRI ArcGIS 10.4.

- Friberg, N., Kronvang, B., Ole Hansen, H., and Svendsen, L.M. (1998). Long-term, habitat-specific response of a macroinvertebrate community to river restoration. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 8, 87–99.
- Frissell, C.A., Liss, W.J., Warren, C.E., and Hurley, M.D. (1986). A hierarchical framework for stream habitat classification: Viewing streams in a watershed context. *Environ. Manage.* 10, 199–214.
- Garshelis, D.L. (2000). Delusions in Habitat Evaluation: Measuring Use, Selection, and Importance. In *Research Techniques in Animal Ecology: Controversies and Consequences*, L. Boitani, and T.K. Fuller, eds. (New York: Columbia University Press), p.
- Hall, L.S., Krausman, P.R., and Morrison, M.L. (1997). The Habitat Concept and a Plea for Standard Terminology. *Wildl. Soc. Bull.* 1973-2006 25, 173–182.
- Hijmans, R.J. (2018). raster: Geographic Data Analysis and Modeling. R package.
- Hughes, A.R., Byrnes, J.E., Kimbro, D.L., and Stachowicz, J.J. (2007). Reciprocal relationships and potential feedbacks between biodiversity and disturbance. *Ecol. Lett.* 10, 849–864.
- Krausman, P.R. (1999). Some Basic Principles of Habitat Use. In *Grazing Behavior of Livestock and Wildlife*, K.L. Launchbaugh, K.D. Sanders, and J.C. Mosley, eds. (Univ. of Idaho, Moscow), p. 6.
- Kristensen, E.A., Thodsen, H., Dehli, B., Kolbe, P.E.Q., Glismann, L., and Kronvang, B. (2013). Comparison of active and passive stream restoration: effects on the physical habitats. *Geogr. Tidsskr.-Dan. J. Geogr.* 113, 109–120.
- Kupilas, B., Hering, D., Lorenz, A.W., Knuth, C., and Gücker, B. (2017). Hydromorphological restoration stimulates river ecosystem metabolism. *Biogeosciences* 14, 1989–2002.
- Lake, P.S. (2003). Ecological effects of perturbation by drought in flowing waters. *Freshw. Biol.* 48, 1161–1172.
- Legendre, P., and Legendre, L. (2012). Complex ecological data sets. In *Developments in Environmental Modelling*, (Elsevier), pp. 1–57.
- Maloney, K.O., and Weller, D.E. (2011). Anthropogenic disturbance and streams: land use and land-use change affect stream ecosystems via multiple pathways. *Freshw. Biol.* 56, 611–626.
- McCain, M., Fuller, D., Decker, L., and Overton, K. (1990). Stream habitat classification and inventory procedures for northern California. USDA For. Serv. Pac. Southwest Reg. FHR Curr. San Franc. CA 94111 1–16.

- Morse, C.C., Huryn, A.D., and Cronan, C. (2003). Impervious Surface Area as a Predictor of the Effects of Urbanization on Stream Insect Communities in Maine, U.S.A. *Environ. Monit. Assess.* *89*, 95–127.
- Newbold, J.D., O'Neill, R.V., Elwood, J.W., and Van Winkle, W. (1982). Nutrient Spiralling in Streams: Implications for Nutrient Limitation and Invertebrate Activity. *Am. Nat.* *120*, 628–652.
- Nimmo, D.G., Mac Nally, R., Cunningham, S.C., Haslem, A., and Bennett, A.F. (2015). Vive la résistance: reviving resistance for 21st century conservation. *Trends Ecol. Evol.* *30*, 516–523.
- Noss, R.F. (1990). Indicators for Monitoring Biodiversity: A Hierarchical Approach. *Conserv. Biol.* *4*, 355–364.
- Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., et al. (2018). *vegan: Community Ecology Package*.
- Palmer, M.A., Hondula, K.L., and Koch, B.J. (2014). Ecological Restoration of Streams and Rivers: Shifting Strategies and Shifting Goals. *Annu. Rev. Ecol. Evol. Syst.* *45*, 247–269.
- Poff, N.L. (1997). Landscape Filters and Species Traits: Towards Mechanistic Understanding and Prediction in Stream Ecology. *J. North Am. Benthol. Soc.* *16*, 391–409.
- R Core Team (2018). *R: A Language and Environment for Statistical Computing* (Vienna, Austria: R Foundation for Statistical Computing).
- Roedenbeck, I., Fahrig, L., Findlay, C.S., Houlihan, J., Jaeger, J., Klar, N., Kramer-Schadt, S., and van der Grift, E. (2007). The Rauischholzhausen Agenda for Road Ecology. *Ecol. Soc.* *12*.
- Shields, F.D., Knight, S.S., Morin, N., and Blank, J. (2003). Response of fishes and aquatic habitats to sand-bed stream restoration using large woody debris. *Hydrobiologia* *494*, 251–257.
- Smith, R.F., Venugopal, P.D., Baker, M.E., and Lamp, W.O. (2015). Habitat filtering and adult dispersal determine the taxonomic composition of stream insects in an urbanizing landscape. *Freshw. Biol.* *60*, 1740–1754.
- Stanley, E.H., Powers, S.M., and Lottig, N.R. (2010). The evolving legacy of disturbance in stream ecology: concepts, contributions, and coming challenges. *J. North Am. Benthol. Soc.* *29*, 67–83.

- Sudduth, E.B., Hassett, B.A., Cada, P., and Bernhardt, E.S. (2011). Testing the Field of Dreams Hypothesis: functional responses to urbanization and restoration in stream ecosystems. *Ecol. Appl.* 21, 1972–1988.
- Templeton, A.R., Brazeal, H., and Neuwald, J.L. (2011). The transition from isolated patches to a metapopulation in the eastern collared lizard in response to prescribed fires. *Ecology* 92, 1736–1747.
- United States Geological Survey (2016). The Streamstats program, online at <http://streamstats.usgs.gov>.
- Waite, I.R. (2014). Agricultural disturbance response models for invertebrate and algal metrics from streams at two spatial scales within the U.S. *Hydrobiologia* 726, 285–303.
- Wang, X., Blanchet, F.G., and Koper, N. (2014). Measuring habitat fragmentation: An evaluation of landscape pattern metrics. *Methods Ecol. Evol.* 5, 634–646.
- Wohl, E., Angermeier, P.L., Bledsoe, B., Kondolf, G.M., MacDonnell, L., Merritt, D.M., Palmer, M.A., Poff, N.L., and Tarboton, D. (2005). River restoration. *Water Resour. Res.* 41, W10301.

CHAPTER 3

MACROINVERTEBRATE ASSEMBLAGE COMPOSITION AND FUNCTIONAL GROUPS UNDER DIFFERENT DISTURBANCE REGIMES: SHORT TERM PATTERNS

ABSTRACT

Streams are dynamic systems shaped by geographic location, hydrology, riparian vegetation, and in-stream habitat. Furthermore, ecosystem disturbance plays a major role in structuring stream communities and ecosystem processes. Disturbances include natural occurrences, such as flooding, drought, and fire events and anthropogenic disturbances such as land use changes, damming, and pollution. Active restoration is intended to ameliorate the effects of anthropogenic disturbance by reintroducing variation and increasing ecosystem heterogeneity, with the goal of shifting the ecosystem into a more diverse and natural state. The act of restoration itself is a pulse disturbance, attempting to shift the community from one system state to another. In this study, I used a Before-After-Control-Impact (BACI) design to investigate the effects of disturbance regimes on compositional and functional attributes of freshwater macroinvertebrate assemblages in a least-impacted stream system, an active flow-through cranberry bog system, and a restored flow-through cranberry bog system in Southeastern Massachusetts from 2014-2017. Overall, I observed a compositional and functional shift in the restored treatment one-year post-restoration, consistent with a pulse disturbance, however after 2 years, the restoration treatment assemblage remained similar to the active cranberry bog.

INTRODUCTION

Global biodiversity is facing decline due to pressure from several synergistic global change drivers (Newbold et al., 2015; Sala et al., 2000). Many of these threats in freshwater systems stem from human activities, particularly due to alterations in land use (Allan, 2004). As a result, freshwater systems have experienced dramatic taxonomic declines, as well as decreased ecosystem services, including nutrient filtration and sediment retention (Dudgeon et al., 2006). In order to address these losses, a deeper understanding of how stressors affect the biota is needed to develop productive strategies to mitigate their loss and recover abrogated functionality (Jones et al., 2018).

Land-use changes and habitat loss are the major drivers of freshwater system decline (Allan, 2004; Sala et al., 2000). The effects these stressors have on the system can be best viewed through the lens of disturbance, an extension of the principles of natural disturbance regimes, which are already thought to largely shape aquatic systems (Lake, 2000; Stanley et al., 2010). Disturbance ecology assesses the effects perturbations have on biomass and the patterns of community response with regard to frequency, intensity, duration, and area of impact (Hughes et al., 2007; Lake, 2000). Related to this are the paired ideas of a community's resistance and resilience, i.e. how much perturbation is required to induce shift in the community and the tendency of the community to return to its initial state, respectively (Folke et al., 2004).

As an extension of these principles, communities can be assessed through the concept of alternative stable states (Beisner et al., 2003). Briefly, this framework posits that multiple states of community composition exist, with varying degrees of resistance and resilience to perturbation. As a result, communities tend toward states that resist

alteration and rarely maintain assemblages that are not self-reinforcing (Folke et al., 2004). Because of this, movement between two stable states requires a perturbation sufficient to overcome the current system's resistance and resilience, so that a reorganization into the new state may occur (Beisner et al., 2003; Stanley et al., 2010).

Restoration of ecosystems attempts to employ this use of alternative stable states to create a shift in community through activities that which have the effect of directed perturbation (Stanley et al., 2010). To determine the result of a restorative action then, the ideal response variable is biodiversity itself. Biodiversity as a simple measure has been used to assess community state following disturbance (Nimmo et al., 2015). However, biodiversity can also be more broadly defined, encompassing hierarchical levels from genetic, taxonomic, and ecosystem diversity, each consisting of compositional, structural, and functional attributes of a community or assemblage (Noss, 1990). This is critically important as a way of measuring shifts in the taxa that make up a community, in addition to changes in richness or evenness that may be unchanged (Christianen et al., 2016) or even increase following disturbance (McCabe and Gotelli, 2000). Assessing communities based on their compositional and/or functional attributes allows us to note both the magnitude of any response to perturbation, as well as make predictions about trending towards alternative states (Avolio et al., 2015).

Freshwater macroinvertebrates (FMI) are a particularly useful assemblage when investigating the impacts of disturbance events. Widely distributed and taxonomically and functionally diverse, FMI represent a clear assemblage to quantify when considering alterations to freshwater habitat (Hauer and Lamberti, 2006). Additionally, FMIs exhibit wide ranges of environmental tolerances, making them differentially susceptible to

perturbations following restoration (Chessman et al., 2007). These shifts have been tied to specific land use alteration studies at both the functional (Fierro et al., 2015; Gutiérrez-Cánovas et al., 2015) and compositional levels (Avolio et al., 2015; Verdonschot, 2009).

Restoration has many goals but ecological restoration to provide habitat and restore biodiversity is a stated goal in at least one-third of literature cited cases (Palmer et al., 2014). As a result, the early phases of restoration and its impact on the community biodiversity is critically important to understanding the timescales and mechanisms by which restorations successfully transition ecosystem states.

STUDY AIM AND HYPOTHESES

The goal of this study was to determine if process-based active restoration (Cranberry Bog Restoration; CBR) resulted in any quantifiable changes to the FMI assemblage, compositionally and functionally, and how those changes compared to control treatments of high (Active Cranberry Bog; ACB) and low (Least-impacted reference; LIR) disturbance. To achieve this goal, I explored 3 primary hypotheses.

First, because the three treatments had different levels of disturbance, I hypothesized that the three treatments would be compositionally distinct preceding the restoration event. Community compositional diversity, e.g. family richness, is highly tied to both water quality and habitat availability (Maul et al., 2004). Impaired systems also are noted to have reduced functional diversity and a more simplified food web structure overall (Hogsden and Harding, 2014). Additionally, release from a press disturbance should allow community diversity to increase per Lake (2003). Thus, if passive restoration marks the

end of a press disturbance, then the initial community at CBR will be distinct from both ACB and the LIR compositionally. Functionally however, ACB and CBR may be similar due to the legacy impacts of farming (Maloney and Weller, 2011).

Secondly, I hypothesized that if cranberry agriculture and the legacy effect of farming represents a biological stress to stream ecosystems (Maloney and Weller, 2011), then initially ACB and CBR will exhibit less overall diversity than LIR but CBR will change through time following restoration, in particular increasing in univariate measures of diversity (e.g. Shannon's diversity), while moving either toward LIR or an alternate assemblage state when examined multi-dimensionally. These responses can be attributed to the creation of new habitat through restoration within the stream, potentially eliminating existing filtering mechanisms, and allowing for colonization from external macroinvertebrate sources (Palmer et al., 2014).

Finally, although I hypothesize initial compositional differences between ACB and CBR and within-study shifts for CRB compositionally, functional change may lag behind and be undetectable within the timeframe of the study. Functional changes, as measured through feeding group distributions, can be driven in large part by food resources (Vannote et al., 1980), and riparian alterations may take on the order of decades to detectably change and thus allow the assemblage to shift on a functional level.

In order to study these predictions, I collected FMI at three treatments (ACB, CBR, and LIR) across three years and analyzed temporal changes through before-after comparison, both in aggregate diversity measures and multidimensional assessments of assemblage composition. My findings suggest that although CBR did experience a compositional shift in the first year post-restoration, this assemblage shift was not

maintained long term and is better understood in the context of a pulse disturbance alteration to the biodiversity of the system. Functional changes were also observed within the first year post-restoration, but similarly were not maintained in the second year.

MATERIALS AND METHODS

Study Area

I conducted this research at three (3) treatments, representing directed pulse disturbance (CBR, Tidmarsh Farms), background disturbance (LIR, Mashpee River), and an agricultural press disturbance (ACB, Active cranberry bog site) in three separate low-order (1st-3rd) streams fed by primarily groundwater and drainage fed ponds in the Atlantic Coastal Pine Barrens Ecoregion of Massachusetts (Figure 1). CBR was taken out of production in 2010, thus entering a phase of passive restoration (Kauffman et al., 1997). CBR then underwent active restoration in Winter 2015/2016. Each treatment consists of three (3) replicate stations defined as 100m length segments where data was collected. Each station was separated by at least 100m in river channel length and sampled during the summer season (between June 21-September 20).

Macroinvertebrate Collection

FMI sampling followed a multi-habitat dip-net protocol (Barbour et al., 1999). Briefly, in-stream macrohabitats were identified to the nearest 5% of stream area as determined via a habitat characterization (McCain et al., 1990) conducted previously. Each 5% increment of habitat constitutes a single collection point, or “jab,” which occurs in a 0.5 square meter area of the macrohabitat. Kicking or jostling with handheld net disturbs

sediments, bank walls, aquatic vegetation, or root masses and a 500µm mesh D-shaped net is swept 5-10 times through the water column, collecting suspended sediments and FMI. All 20 jabs are distributed evenly across the 100m station. The composite collection is transferred into a plastic Nalgene container and preserved with 95% non-denatured ethanol.

FMI were separated from detritus and plant material in the laboratory and placed in fresh 95% ethanol. Invertebrates were examined under magnification, identified to the family level (Merritt et al., 2008; Thorp, 1991) where possible, and counted. Although different life stages were noted for adult and larval Coleopterans, unless taxa only included one life stage in a given treatment/year, life stage was ignored for analysis, as is common in the literature, even in studies specifically addressing taxonomic resolution or habitat specialization (Mueller et al., 2013; Mykrä and Heino, 2017) Full sampled taxa list is available in Appendix A (Tables A.1-A.3). FMI functional feeding group was determined by appropriate references (Hauer and Lamberti, 2006; Merritt et al., 2008).

Statistical Analysis

Community analysis was performed both through time and across treatments. Family level richness, along with Shannon's Diversity, was calculated for each year/treatment pairing and analyzed following effect size for a BACI design:

$$d = (A_I - A_C) - (B_I - B_C)$$

Where *A* represents the mean parameter value in the After period while *B* represents the mean value for the parameter Before (e.g. in 2015, the initial year of study) for both *I*, impact treatment, and *C*, control treatment (Christie et al., 2019).

Confidence intervals (95%) were generated based on these effect sizes and pooled variance:

$$s_p^2 = \frac{(n_{BI} - 1) * s_{BI}^2 + (n_{BC} - 1) * s_{BC}^2 + (n_{AI} - 1) * s_{AI}^2 + (n_{AC} - 1) * s_{AC}^2}{(n_{BI} + n_{BC} + n_{AI} + n_{AC} - 4)}$$

$$CI = d \pm 1.96 * \sqrt{\frac{s_p^2}{n_{BI}} + \frac{s_p^2}{n_{BC}} + \frac{s_p^2}{n_{AI}} + \frac{s_p^2}{n_{AC}}}$$

Where s^2 and n represent the variances and sample size for each set of treatment/period combinations (Christie et al., 2019). This calculation was done for both 2016 and 2017 as separate After periods, as there is no assumption of a stepwise change immediately following restoration combination, comparable to a Before-During-After-Control-Impact study (Roedenbeck et al., 2007). Additionally, due to the differences between control treatments, separate analyses were run treating ACB or LIR as the control treatment. Effect sizes were determined to be statistically significant if the confidence interval did not overlap zero.

In order to assess diversity changes at CBR relative to LIR and ACB conditions, Welch's t-tests were computed pairwise within years, comparing CBR to ACB and CBR to LIR, to determine if CBR's richness or Shannon's diversity altered relative to either reference condition. To increase power without increasing Type I error, pairwise comparisons between LIR and ACB were not conducted. Significance was adjusted due to multiple comparisons using Bonferroni's correction ($\alpha=0.008$, $n=6$) (Legendre and Legendre, 2012).

Additionally, non-metric multi-dimensional scaling (NMDS) using Bray-Curtis dissimilarity, scaled by total sample abundance, was employed to visualize differences in assemblage structure for FMI composition between treatments and years. Differences were evaluated using ANOSIM with year/treatment groupings serving as predictors (*vegan* package, Oksanen et al., 2018). Post hoc tests were unable to be performed on ANOSIM results due to loss of homoscedasticity assumption when data was restricted to subgroups. Key taxonomic groups (i.e. Families) underpinning NMDS separation of treatments were determined based on maximum ranked correlation with NMDS axes, selecting from taxa above median ranked abundance to control for rare taxa artifacts (*ordiselect* function in the *goenveg* package, Goral and Schellenberg, 2018). These families were examined as relative abundance changes through time by treatment, using Welsh's corrected t-tests between years for each treatment with a Bonferroni corrected alpha ($\alpha=0.006$, $n=9$) (Legendre and Legendre, 2012). The same NMDS analysis as described above was also run on feeding group designations in place of taxonomic identities to assess functional differences between year/treatment groupings. All analyses were performed in R (R Core Team, 2018).

RESULTS

Diversity Metrics

An overall examination of simple diversity metrics revealed changes in FMI differently depending on the control treatment. Although LIR was predicted to have the highest familial richness and Shannon's diversity, the differences between treatments was

less than expected. All station taxa richness and Shannon's diversity are available in Appendix A (Tables A.1-A.3).

Richness values at CBR showed a significant negative effect between 2016 and 2015 when compared to ACB, but not LIR ($d_{LIR} = -4.00$ CI: [-16.752,8.753]; $d_{ACB} = -11.000$, CI: [-20.513, -1.487]), (Figure 2A). This may be due to the increase in taxa richness at ACB between 2015 and 2016, whereas no change is seen at LIR (Figure 2B). Within year comparisons by Welsh's t-tests show no significant difference between CBR and ACB or LIR in any year (Welsh's t-tests, Appendix Table B.2) (Figure 2B).

Shannon's Diversity was significantly lower when compared to ACB, but not LIR in both 2016 ($d_{LIR} = 0.194$ CI: [-0.596,0.984]; $d_{ACB} = -1.007$, CI: [-1.693,-0.322]) and 2017 ($d_{LIR} = 0.184$ CI: [-0.446,0.813]; $d_{ACB} = -0.677$, CI: [-1.095,-0.260]) (Figure 3A). As seen for richness, this may be largely due to the increase in ACB Shannon's diversity in 2016 and 2017 (Figure 3B). Within years CBR shows no significant differences with either ACB or LIR (Welsh's t-tests, Appendix B.2) (Figure 3B).

Non-metric Multi-Dimensional Scaling

In addition to univariate metrics as described above, I also examined the compositional distribution of the FMI assemblage through non-metric multi-dimensional scaling. Unexpectedly, there was not a clear separation between CBR and ACB prior to the restoration event, although there was separation between the two and LIR (Figure 4). Changes through time were greatest for CBR's shift between 2015 and 2016 and from 2016 to 2017, as expected would occur following restoration. Although the shift appears the largest between years at CBR, both ACB and LIR exhibit a similar pattern of 2016 shifting away from 2015 and 2017, suggesting a potential regional environmental contributor.

When comparing all years and treatments together, Analysis of Similarity (ANOSIM) demonstrated that between group variability was significantly higher than within group variability ($R: 0.696$, $nperm=999$, $p=0.0001$), demonstrating at a broad scale the differences between treatments and years exceeded station to station variability. Unfortunately, replicate limitations prevent pairwise comparisons, as homoscedasticity is lost when reduced to pairwise comparisons. Thus, I cannot determine significant changes between treatments in a given year or across years within a treatment.

Changes in CBR's 2016 composition appear visually to be driven largely by increases in Hemipteran species (*Corixidae* and *Naucoridae*). In 2017, CBR becomes more similar to its initial 2015 state than either LIR or another alternate state.

Key Family Abundances

To better understand the driving forces behind the observed NMDS separation, the taxonomic groups with the highest ranked correlation to either of the NMDS axes (i.e. best 10%) were obtained. These were further refined by removing those taxa below 50% ranked abundance to avoid the influence of rare taxa. The resulting 4 taxonomic groups were obtained, in order of highest ranked correlation, given in parentheses: *Corixidae* (1), *Limnephilidae* (2), *Hydropsychidae* (4), and *Elmidae* (14). As a result, the changes in CBR can be more clearly seen to be driven by high abundance of *Corixidae* during 2016. Meanwhile changes in *Limnephilidae*, *Elmidae*, and *Hydropsychidae* show clear taxonomic drivers that separate LIR from both ACB and CBR.

When examining *Corixidae* abundance patterns, although no significant changes occur across time, there is a large visible increase in 2016 that dissipates in 2017 (Figure 5A). The high variability in 2016 combined with the limited sample size of stations limits

the ability to determine if this shift is significant (2015-2016: Welsh's $t=-3.933$, $v=2.009$, $p=0.059$).

For *Limnephilidae*, we see shifts in abundance that largely occur in LIR, with higher abundances in both 2016 and 2017 compared to 2015 (Figure 5B). Due to high station variability, this change is not statistically significant (2015-2017: Welsh's $t=-5.931$, $v=2.578$, $p=0.0145$). Although there are year-to-year changes at LIR, notably this taxa has almost no abundance at either ACB or CBR at any year. Thus, *Limnephilidae* aids in explaining primarily the separation of LIR from ACB or CBR observable in the NMDS.

Elmidae, as combined larvae and adult abundance, also show the separation of LIR from ACB and CBR (Figure 5C). Although the changes in relative abundance are primarily between years at LIR, opposite of those seen for *Limnephilidae*, the high station-to-station variability means that none of these changes are statistically significant (Welsh's t-tests, Appendix Table C.1).

Finally, although again there are no significant changes associated with *Hydropsychidae* abundance (Welsh's t-tests, Appendix Table C.1), the high proportion of *Hydropsychid* individuals found at LIR stations again underpins the separation of this treatment from the two treatments impacted by agriculture (Figure 5D).

Functional Feeding Groups

Although compositional shifts occurred following the restoration activities in 2015, functional changes were hypothesized to lag. However, when examining functional feeding group breakdown, a similar pattern appears, with a shift in 2016 CBR towards more piercing herbivores (Figure 6).

This pattern also appears visually to be driven by the increase in *Piercing Herbivores* (which include *Corixidae*) during 2016. Although treatment/year groupings are statistically significant different from between group distances (ANOSIM, R: 0.6189, nperm=999, p=0.001), post-hoc tests are again unable to be computed due to low station replication. However, visually, the clear separation for 2016 CBR can be seen.

DISCUSSION

Overall, the results of my study lead to four main findings. First, aggregate metrics of biodiversity demonstrated unexpected similarities across all treatments, which was not matched by examination of the assemblage as a whole and its changes through time. Secondly, rather than distinct initial compositions for all three treatments as expected, CBR and ACB shared a very similar FMI assemblage before the restoration. Thirdly, restoration of the former cranberry bog at CBR coincided with a one-year shift in composition and function driven primarily by a single taxon, which was not maintained in the following year. Finally, compositional shifts at CBR and across all years at LIR is driven in large part by a few key taxa with important life histories.

Diversity Metrics

The richness and Shannon's Diversity measures broadly show similarity in treatments across time. There are increases in taxa richness and Shannon's Diversity through time for both ACB and CBR, but no consistent pattern of that matches my expectation of LIR having higher richness or diversity, given its lack of agricultural impact at our study scale (Maloney and Weller, 2011). However, simplified diversity metrics can

miss compositional differences between samples (Christianen et al., 2016). Additionally, the taxonomic resolution may limit the discriminatory ability of these univariate metrics (Chessman et al., 2007). Thus, while important metrics of assessing assemblage changes, investigation into the taxonomic drivers of treatment to treatment differences is merited, especially in cases with limited sample size and short study windows.

Compositional and Functional Assemblage Attributes Shifts

Investigating the assemblage data through NMDS reveals more fine-grained differences between treatments and through time. To begin with, my expectation that all three treatments would inhabit unique compositional space in 2015 was incorrect, with CBR and ACB exhibiting very similar composition. I expected this initial difference due to the land use associated with each treatment, with agriculture acting as a press disturbance and passive recovery at CBR resulting in a less managed assemblage (Harding et al., 1998; Wohl et al., 2005). However, other studies have shown legacy impacts that affect FMI assemblages longer than the passive phase of restoration at CBR (Engel, 2010).

Secondly, although compositional shifts do occur during the study, CBR's post-restoration assemblages fit more of the pattern expected by a pulse disturbance – with a large initial shift followed by a recovery toward the initial state (Lake, 2000). Although restorations often do occur with “reorganization phases” with higher variability (Stanley et al., 2010), the highest variability within our study is present within LIR rather than post-restoration CBR. I expect it FMI assemblage composition may not shift away from pre-restoration until riparian regrowth occurs, shifting carbon inputs in the system.

All three treatments do show some yearly variation that distinguishes 2016 from either 2015 or 2017. While this coincides with the restoration as an explanation for CBR's

shifts, 2016 was also a drought year including the Southeast and Cape regions of Massachusetts (Department of Conservation and Recreation). The effects of the drought may exacerbate differences seen at CBR, although its magnitude in shift is not matched in either ACB or LIR, suggesting drought does not fully explain the observed differences.

When examined functionally, i.e. through functional feeding groups, I expected initial FMI similarities between CBR and ACB to exist based on the legacy of farming affecting available habitat (Harding et al., 1998). Surprisingly, I found that all three treatments were similar to each other, with the notable exception of LIR in 2016. In particular, I expected LIR to be higher in relative abundance of shredder species, given the lack of canopy cover, and associated deciduous leaf litter, at ACB and CBR treatments. On-site canopy cover can greatly influence the local macroinvertebrate community by providing necessary material for shredder species (Violin et al., 2011). However, functional similarities between treatment assemblages may also be related to shared characteristics inherent to all 3 treatments, such as their status as wadable coastal streams downstream of lakes. The largest change observable in the functional NMDS is the 2016 CBR shift towards a high proportion of piercing herbivores, discussed above. However, it is worth mentioning that NMDS is sensitive to input data, and underlying differences between the three treatments may be obscured by the larger change in CBR over time. The limitations of small sample size prevent pairwise comparisons to address these issues.

Finally, the individual taxa underlying these multi-dimensional differences between treatments and across years can be determined and analyzed separately. The four taxa with the highest axes fit and with above median ranked abundance for the NMDS compositional data are *Corixidae*, *Limnephilidae*, *Elmidae*, and *Hydropsychidae*.

Corixidae data shows the underpinning of the CBR shifts and themselves are known to be indicators of lentic quality (Merritt et al., 2008). Within the *Corixidae* family, there also exists a large variation in across land colonization ability, suggesting the increase at CBR may be tied to dispersal ability of sub-family taxa (Brown, 1951). Meanwhile, the other three taxa demonstrate the major differences between LIR and CBR/ACB, given their abundances at LIR. *Elmidae* and *Hydropsychidae* in particular are associated with high water flow and high oxygenation (Hauer and Lamberti, 2006; Merritt et al., 2008). The alternation between 2015 and 2016/2017 for *Limnephilidae* and *Elmidae* abundance may also be a result of drought effects in 2016, with *Limnephilidae* taxa tending to be flow generalists (Merritt et al., 2008). However, the increase in *Hydropsychidae*, also a rheophilic taxa (Hauer and Lamberti, 2006), suggests flow changes were not limiting in 2016. Thus, we can see that although the CBR shifts are driven by the transitory abundance of a lentic specialist immediately following restoration, the LIR differences are driven by taxa requiring high water flow and oxygenation.

Given the diversity that exists below the family level, it is important to keep in mind that increased taxonomic specificity may provide more information about abiotic and biotic drivers at these treatments (Arscott et al., 2006). At the same time, fine grain differentiation between treatments may lead to overemphasis on site-specific differences driven by taxa occupying fundamentally similar niches and thus ignore important similarities.

CONCLUSIONS

The influence of disturbance regimes on community biodiversity is a well-researched field for many forms of disturbance (Lake, 2000) and potential interactive effects (Hughes et al., 2007). However, the broader context of restoration as a disturbance its impact on the community, either through the cessation of an external pressure (Mateos-Molina et al., 2014) or through active restoration means (Stanley et al., 2010) remains a fruitful area of exploration. In particular, as restorations become increasingly used to rehabilitate freshwater systems impaired by land-use changes, the alterations to the in-stream assemblage, both compositionally and functionally, become critical to better understanding the course of recovery and whether alternate assemblage states are being achieved.

Through my study, I found that the initial phase of an intensive restoration exhibits all the hallmarks of a pulse type disturbance, including recovery toward the initial pre-restoration state. While this has been shown in other studies, these are often restricted to simplified diversity metrics within the first few years (Friberg et al., 1998). These metrics detail changes in overall richness or evenness, but do not provide information about taxa turnover, which may provide insights into the processes occurring in the “reorganization phase” (Stanley et al., 2010). Post-restoration, CBR experienced a dramatic increase in a lentic-preferring species of herbivore, potentially related to the flow alterations and macrophyte growth in new channel, in contrast to the rheophilic taxa present at LIR. To avoid this pulse, future restoration projects may benefit from a design that either transitions new channel to desired flow velocity earlier or manages macrophytic growth.

The long-term nature of restoration projects requires longer term investigation of the trends following the restructuring actions. As a result, the findings in the near-term raise questions about their stability through time, as reforestation of riparian areas occurs. Many studies have identified the increase in variability as a hallmark of early restoration (Carpenter and Brock, 2006; Stanley et al., 2010). Thus, even without strong evidence of shifts within the first two years, the shifts away from the initial state may occur over the next several years, making long-term study through the integrative lens of biodiversity essential for deeper understanding of restoration's progress and how to improve these projects in the future.

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FIGURES

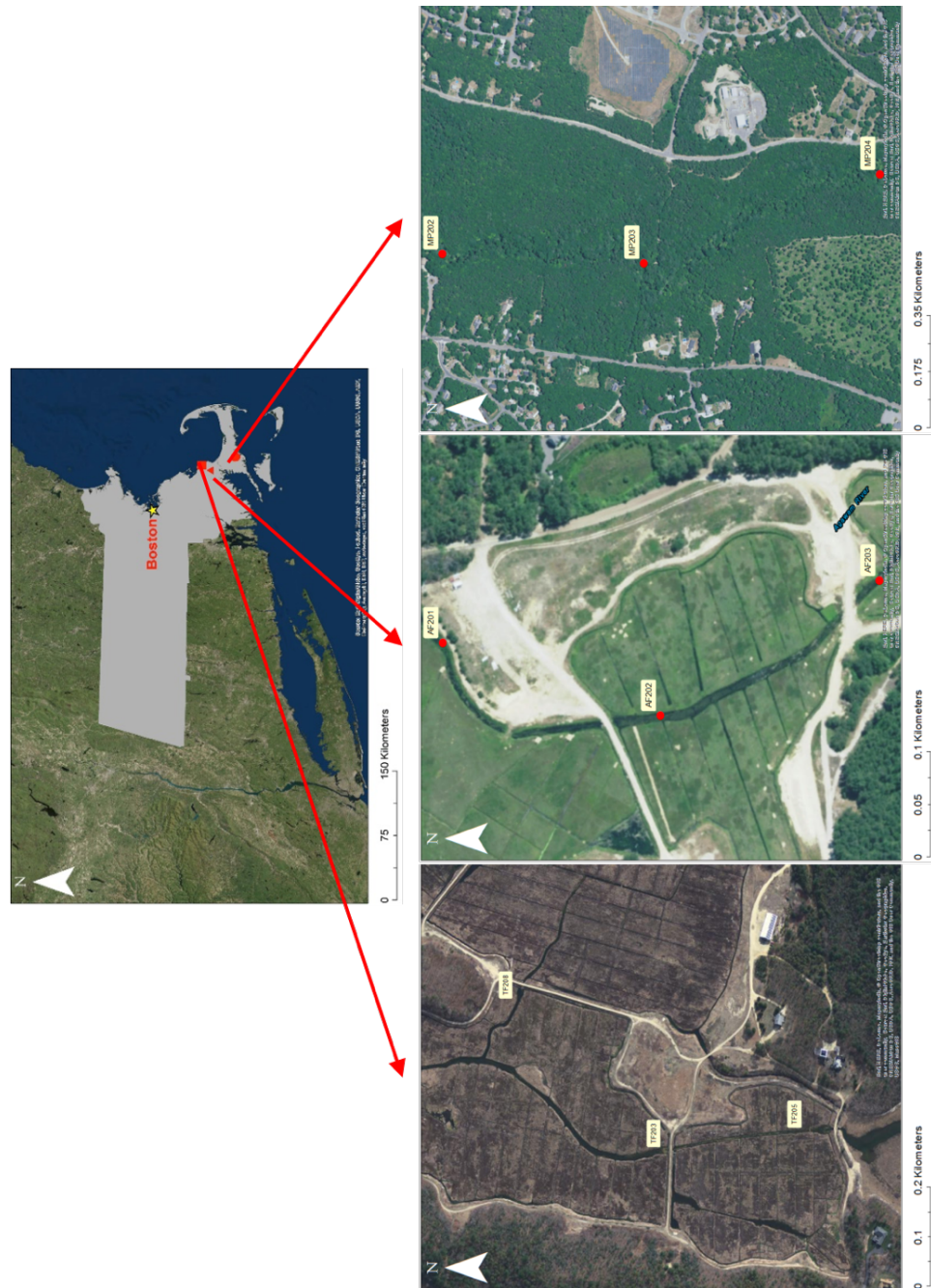


Figure 1: Experimental treatment study locations and their relative locations in Southeastern Massachusetts. From left to right: Tidmarsh Farms (CBR, pre-restoration imagery), an active flow-through cranberry bog (ACB), and the Mashpee River flowing through the Trustees of Reservations conservation land (LIR) (ArcGIS, ESRI). Map created by S. McCanty.

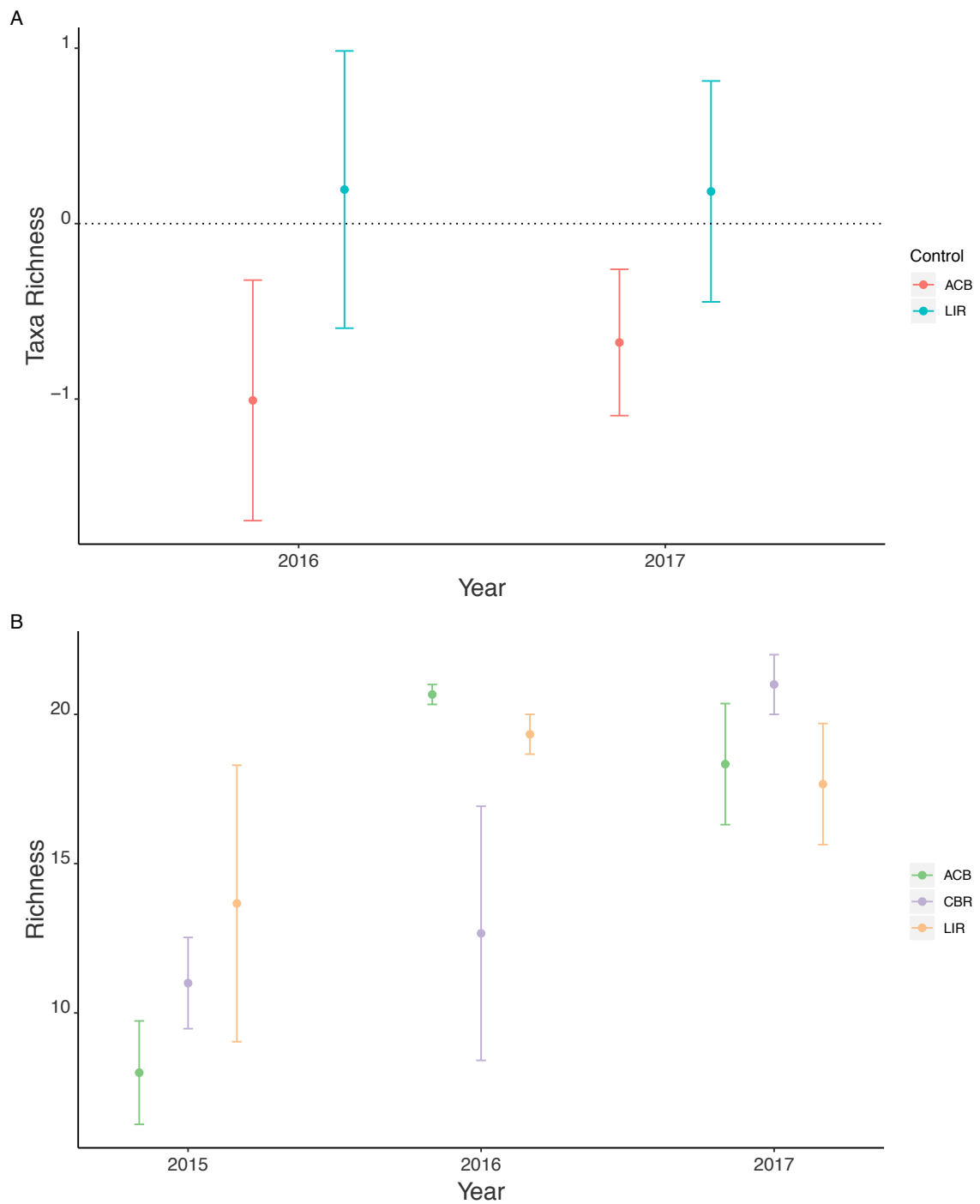


Figure 2: Taxa Richness as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests ($\alpha=0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

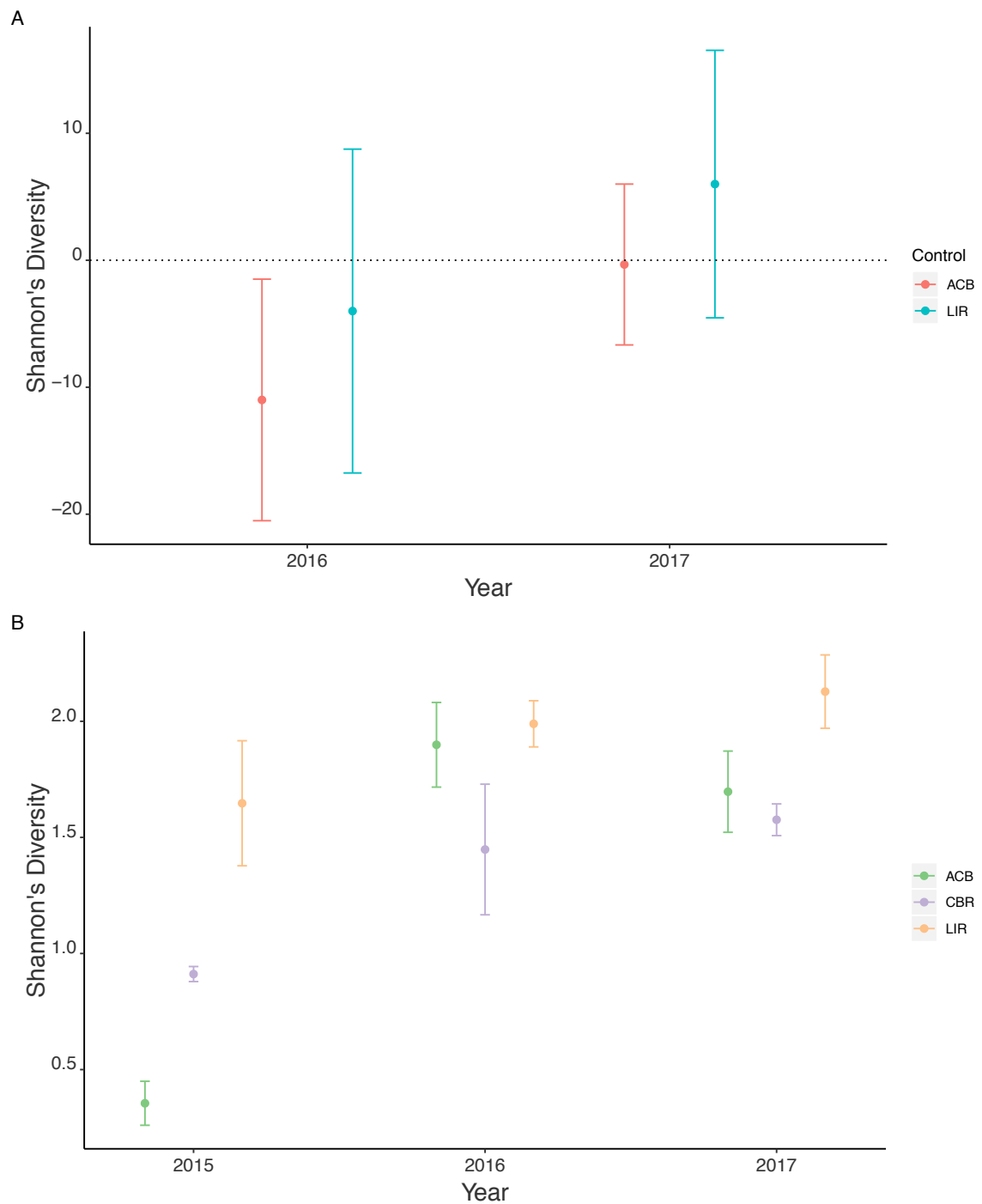


Figure 3: Shannon's Diversity as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence intervals not overlapping zero (A) or pair-wise Welch's t-tests (Bonferroni-corrected $\alpha=0.008$) between CBR and other treatments within year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

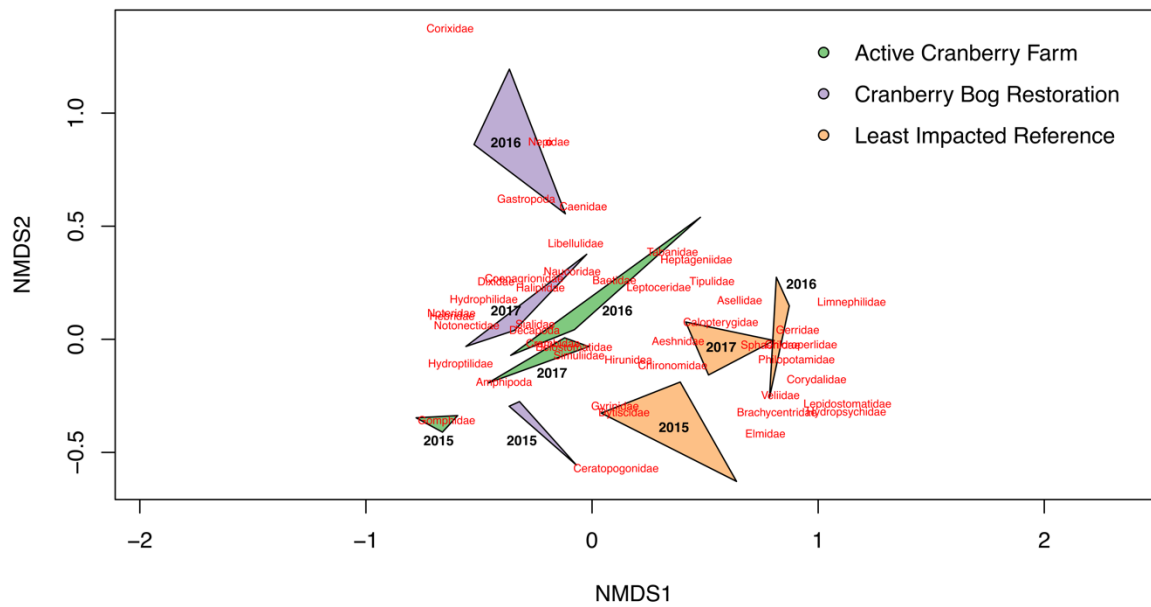


Figure 4: Non-metric Multi-Dimensional Scaling (NMDS) plot of Summer freshwater macroinvertebrate assemblages from 2015-2017, at each of the treatment locations. Polygons bound all stations within each treatment/year. Taxonomic groups used to generate the NDMS are overlaid on the visualization. Stress = 0.145.

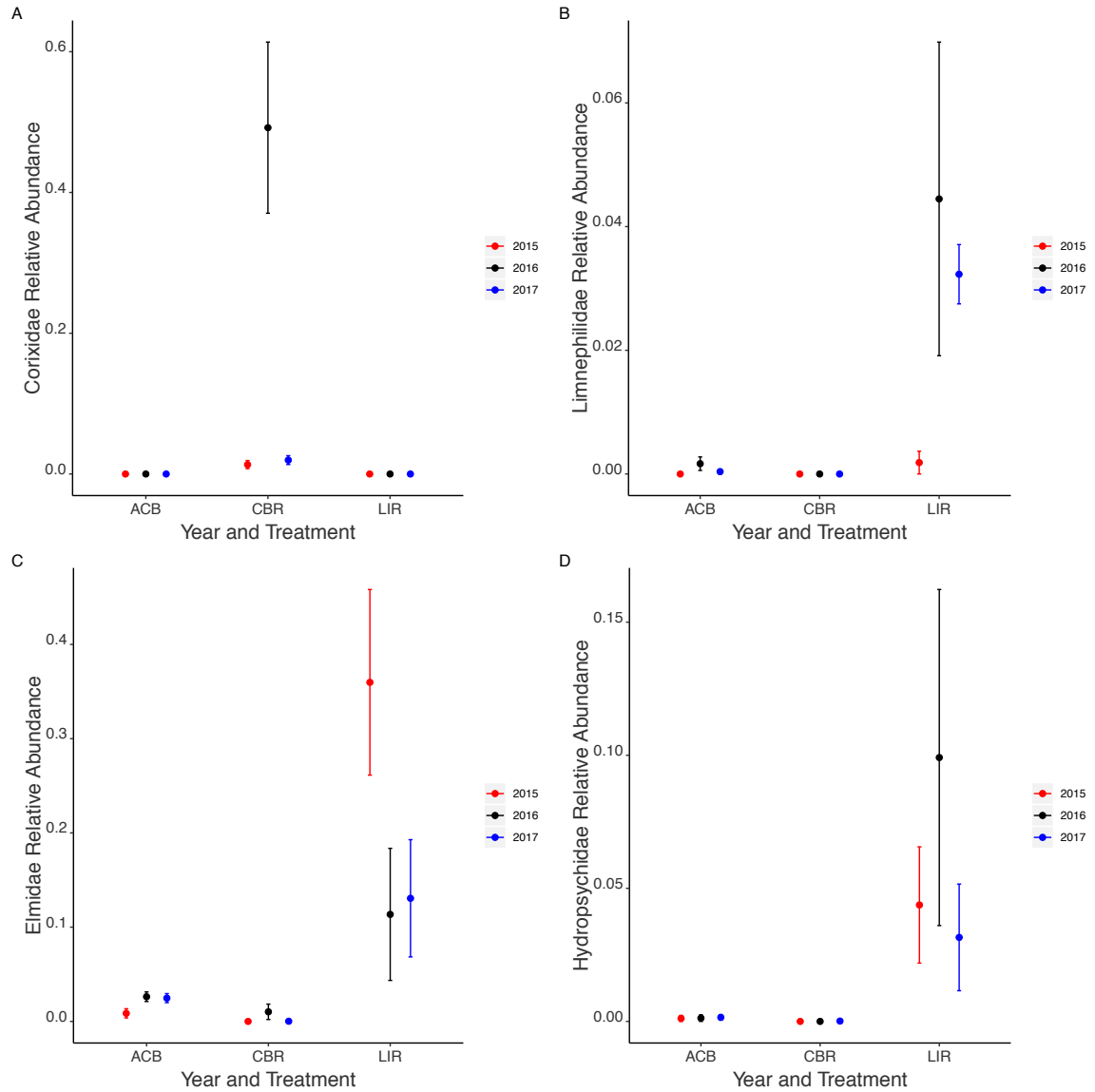


Figure 5: Select family relative abundances at each treatment/year. Families were selected from the top 10% of NMDS axes ranked correlation and above 50% relative rank abundance. *Corixidae* (A), *Limnephilidae* (B), *Elmidae* (C), and *Hydropsychidae* (D). Within year differences for each treatment were tested via two tailed Welch's t-tests (Bonferroni-adjusted $\alpha=0.006$). No significant relationships were found.

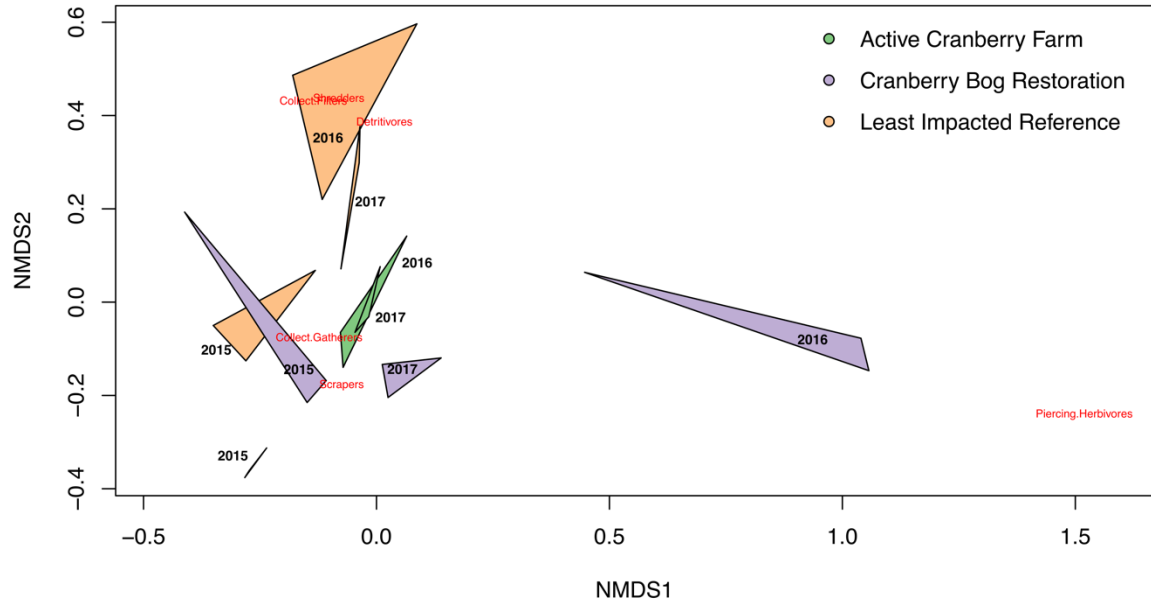


Figure 6: NMDS based on Functional Feeding Group relative abundances. Taxonomic families were converted to functional feeding groups based on literature associations. Due to the diversity within taxa, individual families were not limited to one functional feeding group. Stress = 0.085.

APPENDIX A

Table A.1: Summer 2015 FMI raw collection data with total sample abundance and univariate diversity metrics listed below.

Taxa	2015 ACB			2015 LIR			2015 CBR		
	201	202	203	201	202	203	203	205	208
<i>Aeshnidae</i>	0	0	0	0	1	9	0	1	0
<i>Amphipoda</i>	276	403	185	181	25	160	1143	424	629
<i>Asellidae</i>	0	4	2	21	0	78	17	3	22
<i>Baetidae</i>	0	0	0	0	0	16	3	0	0
<i>Belostomatidae</i>	0	0	0	0	0	0	0	0	0
<i>Beraeidae</i>	0	0	0	0	0	0	0	0	0
<i>Brachycentridae</i>	0	0	0	0	0	8	0	0	0
<i>Caenidae</i>	0	0	0	0	0	0	0	0	0
<i>Calopterygidae</i>	0	0	0	13	0	12	0	0	0
<i>Ceratopogonidae</i>	0	0	0	0	0	0	3	3	0
<i>Chironomidae</i>	5	25	1	15	13	47	271	629	169
<i>Chloroperlidae</i>	0	0	0	0	0	12	0	0	0
<i>Coenagrionidae</i>	0	3	0	0	0	0	1	1	0
<i>Corixidae</i>	0	0	0	0	0	0	38	7	8
<i>Corydalidae</i>	0	0	0	0	0	3	0	0	0
<i>Crambidae</i>	0	0	0	0	0	0	0	0	0
<i>Dixidae</i>	0	0	0	0	0	0	0	0	0
<i>Dytiscidae</i> ¹	0	0	0	0	0	11	17	15	1
<i>Decapoda</i>	0	0	1	0	0	0	0	0	0
<i>Elmidae</i> ¹	5	4	0	92	64	155	0	0	0
<i>Gomphidae</i>	1	4	0	0	0	1	0	0	0
<i>Gastropoda</i>	0	1	2	1	0	3	9	0	2
<i>Gyrinidae (Adult)</i>	0	0	0	0	0	1	0	0	0
<i>Gerridae</i>	0	0	0	0	0	2	0	0	0
<i>Haliplidae</i> ¹	0	0	0	0	0	13	25	3	14
<i>Heptageniidae</i>	6	0	0	9	2	0	0	0	0
<i>Hirunidea</i>	2	2	0	9	0	0	0	9	66
<i>Hydrophilidae</i>	0	0	0	2	0	0	3	1	0
<i>Hydropsychidae</i>	1	0	0	6	10	16	0	0	0
<i>Hydroptilidae</i>	0	0	0	0	0	0	0	0	0
<i>Hebridae</i>	0	0	0	0	0	0	0	0	0
<i>Lepidostomatidae</i>	0	0	0	0	0	7	0	0	0
<i>Leptoceridae</i>	0	1	0	0	0	1	0	0	0
<i>Libellulidae</i>	0	0	0	0	0	0	0	0	0

<i>Limnephilidae</i>	0	0	0	2	0	0	0	0	0
<i>Nepidae</i>	0	0	0	0	0	0	0	0	0
<i>Naucoridae</i>	0	0	0	0	0	0	0	0	0
<i>Noteridae (Adult)</i>	0	0	0	0	0	0	0	0	0
<i>Notonectidae</i>	0	0	0	0	0	0	0	0	0
<i>Philopotamidae</i>	0	0	0	0	0	0	0	0	0
<i>Sialidae</i>	0	0	0	0	0	0	0	0	0
<i>Simuliidae</i>	1	1	0	0	0	13	0	0	0
<i>Sphaeriidae</i>	0	0	0	7	0	6	1	0	0
<i>Tabanidae</i>	0	0	0	0	0	0	0	0	0
<i>Tipulidae</i>	0	2	0	4	0	3	2	1	0
<i>Veliidae</i>	0	0	0	0	0	0	0	0	0
Total Abundance	297	450	191	362	115	577	1533	1097	911
Taxa Richness	8	11	5	13	6	22	13	12	8
Shannon's Diversity	0.376	0.508	0.181	1.563	1.228	2.150	0.869	0.891	0.975

¹Taxa were found as both larvae and adults at each treatment present and thus were combined for all analyses.

Table A.2: Summer 2016 FMI raw collection data with total sample abundance and univariate diversity metrics listed below.

Taxa	2016 ACB			2016 LIR			2016 CBR		
	201	202	203	202	203	204	211	212	213
<i>Aeshnidae</i>	17	14	10	5	4	5	0	0	1
<i>Amphipoda</i>	120	835	316	81	122	64	52	71	323
<i>Asellidae</i>	30	36	48	48	363	83	5	13	126
<i>Baetidae</i>	36	43	8	3	3	14	0	19	15
<i>Belostomatidae</i>	0	0	0	0	0	0	0	0	0
<i>Beraeidae</i>	0	0	0	0	0	0	0	0	1
<i>Brachycentridae</i>	0	0	0	3	0	0	0	0	0
<i>Caenidae</i>	0	0	0	0	0	0	0	0	0
<i>Calopterygidae</i>	42	0	11	5	1	2	0	0	0
<i>Ceratopogonidae</i>	0	0	0	1	0	0	0	0	0
<i>Chironomidae</i>	121	75	99	52	153	231	4	2	68
<i>Chloroperlidae</i>	0	0	0	4	9	14	0	0	0
<i>Coenagrionidae</i>	61	59	36	0	0	0	5	6	38
<i>Corixidae</i>	0	0	0	0	0	0	133	284	294
<i>Corydalidae</i>	0	0	0	5	3	0	0	0	1
<i>Crambidae</i>	7	7	5	0	0	0	0	0	1
<i>Dixidae</i>	0	0	0	0	0	0	0	0	0
<i>Dytiscidae</i> ¹	0	3	0	0	0	2	0	0	1
<i>Decapoda</i>	5	9	5	0	0	0	0	0	0
<i>Elmidae</i> ¹	28	24	21	131	53	14	0	2	31
<i>Gomphidae</i>	0	0	3	0	0	0	0	0	0
<i>Gastropoda</i>	18	85	40	0	0	1	10	73	190
<i>Gyrinidae (Adult)</i>	0	0	0	0	0	0	0	0	0
<i>Gerridae</i>	0	0	1	0	1	2	0	0	0
<i>Haliplidae</i> ¹	0	2	1	0	0	0	0	7	18
<i>Heptageniidae</i>	263	55	8	6	3	0	0	0	0
<i>Hirunidea</i>	0	8	2	8	16	13	1	0	8
<i>Hydrophilidae</i>	1	1	1	0	0	0	0	2	1
<i>Hydropsychidae</i>	3	0	0	117	23	25	0	0	0
<i>Hydroptilidae</i>	0	1	0	0	0	0	0	0	0
<i>Hebridae</i>	2	0	0	0	0	0	0	0	0
<i>Lepidostomatidae</i>	0	0	0	15	0	0	0	0	0
<i>Leptoceridae</i>	5	2	1	1	2	0	0	0	0
<i>Libellulidae</i>	0	0	0	0	1	0	0	0	7
<i>Limnephilidae</i>	3	0	1	0	38	49	0	0	0

<i>Nepidae</i>	0	0	0	0	0	0	0	0	1
<i>Naucoridae</i>	0	0	0	0	0	0	0	0	3
<i>Noteridae (Adult)</i>	0	0	0	0	0	0	0	0	0
<i>Notonectidae</i>	0	0	0	0	0	0	0	0	0
<i>Philopotamidae</i>	0	0	0	4	6	1	0	0	0
<i>Sialidae</i>	4	4	0	0	0	0	0	0	0
<i>Simuliidae</i>	16	134	169	14	7	25	0	0	0
<i>Sphaeriidae</i>	0	0	0	10	16	6	0	0	6
<i>Tabanidae</i>	9	2	1	0	0	0	0	0	0
<i>Tipulidae</i>	15	0	0	7	9	7	0	0	42
<i>Veliidae</i>	0	0	0	0	0	0	0	0	0
Total Abundance	806	1399	787	520	833	558	210	479	1176
Taxa Richness	21	20	21	20	20	18	7	10	21
Shannon's Diversity	2.229	1.594	1.877	2.164	1.82	1.983	1.059	1.291	1.994

¹Taxa were found as both larvae and adults at each treatment present and thus were combined for all analyses.

Table A.3: Summer 2017 FMI raw collection data with total sample abundance and univariate diversity metrics listed below.

Taxa	2017 ACB			2017 LIR			2017 CBR		
	201	202	203	202	203	204	211	212	213
<i>Aeshnidae</i>	6	8	11	4	1	0	2	2	1
<i>Amphipoda</i>	547	104	373	82	32	42	482	565	2852
<i>Asellidae</i>	9	8	26	15	55	25	34	75	420
<i>Baetidae</i>	48	4	35	8	4	1	10	15	8
<i>Belostomatidae</i>	0	1	1	0	0	0	0	0	1
<i>Beraeidae</i>	0	0	0	0	0	0	0	0	0
<i>Brachycentridae</i>	0	0	0	0	0	0	0	0	0
<i>Caenidae</i>	0	0	0	0	0	0	0	6	0
<i>Calopterygidae</i>	1	0	9	7	3	1	0	0	0
<i>Ceratopogonidae</i>	0	0	0	0	0	0	0	0	2
<i>Chironomidae</i>	36	59	96	35	63	54	23	298	387
<i>Chloroperlidae</i>	0	0	0	17	3	8	0	0	0
<i>Coenagrionidae</i>	11	15	47	0	0	0	95	75	280
<i>Corixidae</i>	0	0	0	0	0	0	25	28	59
<i>Corydalidae</i>	0	0	0	2	0	0	0	0	0
<i>Crambidae</i>	14	26	10	2	0	0	20	4	10
<i>Dixidae</i>	0	0	0	0	0	0	1	4	8
<i>Dytiscidae¹</i>	0	0	0	0	1	0	0	3	28
<i>Decapoda</i>	2	1	4	0	0	0	0	0	0
<i>Elmidae¹</i>	14	7	26	82	35	3	0	1	0
<i>Gomphidae</i>	4	0	1	0	0	0	0	0	0
<i>Gastropoda</i>	11	5	36	2	0	1	42	682	552
<i>Gyrinidae (Adult)</i>	1	0	0	0	0	0	0	0	0
<i>Gerridae</i>	0	0	0	0	0	0	0	0	0
<i>Haliplidae¹</i>	0	1	0	0	0	0	6	41	59
<i>Heptageniidae</i>	7	0	70	18	3	0	2	0	0
<i>Hirunidea</i>	0	0	2	3	3	4	1	34	107
<i>Hydrophilidae</i>	0	0	0	0	0	0	11	6	17
<i>Hydropsychidae</i>	1	0	3	6	18	1	0	0	2
<i>Hydroptilidae</i>	0	0	0	0	0	0	0	0	0
<i>Hebridae</i>	0	0	0	0	0	0	18	8	45
<i>Lepidostomatidae</i>	0	0	0	0	3	0	0	0	0
<i>Leptoceridae</i>	0	1	1	0	0	0	0	8	0
<i>Libellulidae</i>	0	1	0	0	0	0	1	0	1
<i>Limnephilidae</i>	0	0	1	8	9	6	0	0	0

<i>Nepidae</i>	0	0	0	0	0	0	0	0	0
<i>Naucoridae</i>	0	1	0	0	0	0	0	0	0
<i>Noteridae (Adult)</i>	0	0	0	0	0	0	2	2	7
<i>Notonectidae</i>	0	0	0	0	0	0	0	0	1
<i>Philopotamidae</i>	0	0	0	27	5	6	0	0	0
<i>Sialidae</i>	8	0	2	0	0	0	0	0	0
<i>Simuliidae</i>	204	0	108	15	9	2	1	0	0
<i>Sphaeriidae</i>	0	0	0	2	2	0	0	13	11
<i>Tabanidae</i>	1	0	2	1	0	0	1	0	0
<i>Tipulidae</i>	0	0	1	1	4	2	1	0	32
<i>Veliidae</i>	0	0	0	13	0	0	0	0	0
Total Abundance	925	242	865	350	253	156	778	1870	4890
Taxa Richness	18	15	22	21	18	14	20	20	23
Shannon's Diversity	1.378	1.731	1.981	2.364	2.192	1.828	1.495	1.712	1.522

¹Taxa were found as both larvae and adults at each treatment present and thus were combined for all analyses.

APPENDIX B

Table B.1: Effect size (BACI) of change before and after restoration (2015 to 2016, or 2015 to 2017) relative to control (LIR or ACB) treatment. Each effect size is calculated as described in-text, with 95% confidence intervals generated from pooled variance. Bolding represents effects with confidence intervals not overlapping 0. ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

Year	Control	Taxa Richness		Shannon's Diversity	
		d	95% CI	d	95% CI
2016	LIR	-4.000	[-16.752,8.753]	0.194	[-0.596,0.984]
2016	ACB	-11.000	[-20.513,-1.487]	-1.007	[-1.693,-0.322]
2017	LIR	6.000	[-4.535,16.535]	0.184	[-0.446,0.813]
2017	ACB	-0.333	[-6.668,6.001]	-0.677	[-1.095,-0.260]

Table B.2: Within year differences between CBR and control treatments. T values represent Welch's two-tailed t-tests between CBR and listed treatment in the same year. Due to unequal variances, Welch's t-tests calculate v , a modified df. Bolding represents changes that are significantly different ($\alpha = 0.008$, Bonferroni correction for 6 comparisons per year). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

Year	CBR:Treatment	Taxa Richness			Shannon's Diversity		
		T value	P value	v	T value	P value	v
2015	ACB	1.299	0.265	3.938	5.560	0.019	2.461
	LIR	-0.547	0.631	2.430	-2.712	0.110	2.058
2016	ACB	-1.874	0.200	2.025	-1.345	0.261	3.429
	LIR	-1.548	0.256	2.098	-1.815	0.186	2.492
2017	ACB	1.180	0.325	2.919	-0.644	0.572	2.596
	LIR	1.474	0.232	2.919	-3.207	0.056	2.723

APPENDIX C

Table C.1: Between year differences for all treatments in relative abundance of each key Family group. T-values represent Welsh's two-tailed t-tests within treatment in the given years. Due to unequal variances, Welsh's t-tests calculate ν , a modified df. Bolding represents changes that are significantly different ($\alpha=0.006$, Bonferroni correction for 9 comparisons per taxa). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

Treatment	Years	<u><i>Corixidae</i></u>			<u><i>Limnephilidae</i></u>		
		T value	P value	ν	T value	P value	ν
ACB	2015-2016	-	-	-	-1.524	0.267	2.000
	2016-2017	-	-	-	1.104	0.365	2.490
	2015-2017	-	-	-	-1.000	0.423	2.000
CBR	2015-2016	-3.933	0.059	2.009	-	-	-
	2016-2017	3.874	0.060	2.011	-	-	-
	2015-2017	-0.752	0.494	3.974	-	-	-
LIR	2015-2016	-	-	-	-1.677	0.234	2.021
	2016-2017	-	-	-	0.472	0.681	2.143
	2015-2017	-	-	-	-5.931	0.014	2.578

Table C.1 continued

Treatment	Years	<u><i>Elmidae</i></u>			<u><i>Hydropsychidae</i></u>		
		T value	P value	ν	T value	P value	ν
ACB	2015-2016	-2.505	0.067	3.992	-0.071	0.947	3.960
	2016-2017	0.213	0.842	3.987	-0.171	0.873	3.862
	2015-2017	-2.362	0.078	3.999	-0.259	0.808	3.967
CBR	2015-2016	-1.244	0.339	2.000	-	-	-
	2016-2017	1.222	0.346	2.002	-1.000	0.423	2.000
	2015-2017	-1.000	0.423	2.000	-1.000	0.423	2.000
LIR	2015-2016	2.037	0.119	3.612	-0.829	0.479	2.472
	2016-2017	-0.182	0.864	3.944	1.020	0.399	2.399
	2015-2017	1.967	0.133	3.377	0.411	0.702	3.970

LITERATURE CITED

- Allan, J.D. (2004). Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems. *Annu. Rev. Ecol. Evol. Syst.* 35, 257–284.
- Arscott, D.B., Jackson, J.K., and Kratzer, E.B. (2006). Role of rarity and taxonomic resolution in a regional and spatial analysis of stream macroinvertebrates. *J. North Am. Benthol. Soc.* 25, 977–997.
- Avolio, M.L., Pierre, K.J.L., Houseman, G.R., Koerner, S.E., Grman, E., Isbell, F., Johnson, D.S., and Wilcox, K.R. (2015). A framework for quantifying the magnitude and variability of community responses to global change drivers. *Ecosphere* 6, 1–14.
- Barbour, M.T., Gerritsen, J., Snyder, B.D., and Stribling, J.B. (1999). Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, (Washington, D.C.: U.S. Environmental Protection Agency; Office of Water).
- Beisner, B.E., Haydon, D.T., and Cuddington, K. (2003). Alternative Stable States in Ecology. *Front. Ecol. Environ.* 1, 376–382.
- Brown, E.S. (1951). The relation between migration-rate and type of habitat in aquatic insects, with special reference to certain species of Corixidae. *Proc. Zool. Soc. Lond.* 121, 539–545.
- Carpenter, S.R., and Brock, W.A. (2006). Rising variance: a leading indicator of ecological transition. *Ecol. Lett.* 9, 311–318.
- Chessman, B., Williams, S., and Besley, C. (2007). Bioassessment of streams with macroinvertebrates: effect of sampled habitat and taxonomic resolution. *J. North Am. Benthol. Soc.* 26, 546–565.
- Christianen, M., van der Heide, T., Holthuijsen, S., van der Reijden, K., Borst, A., and Olff, H. (2016). Biodiversity and food web indicators of community recovery in intertidal shellfish reefs. *Biol. Conserv.*
- Christie, A.P., Amano, T., Martin, P.A., Shackelford, G.E., Simmons, B.I., and Sutherland, W.J. (2019). Simple study designs in ecology produce inaccurate estimates of biodiversity responses. *J. Appl. Ecol.* 56, 2742–2754.
- Department of Conservation and Recreation Past Drought Declarations Maps and History.

- Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z.-I., Knowler, D.J., Lévêque, C., Naiman, R.J., Prieur-Richard, A.-H., Soto, D., Stiassny, M.L.J., et al. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. *Biol. Rev.* 81, 163–182.
- Engel, H. (2010). Stream quality among active and restoring river-based cranberry bogs. Unpubl. MBL Woods Hole.
- Fierro, P., Bertrán, C., Mercado, M., Peña-Cortés, F., Tapia, J., Hauenstein, E., Caputo, L., and Vargas-Chacoff, L. (2015). Landscape composition as a determinant of diversity and functional feeding groups of aquatic macroinvertebrates in southern rivers of the Araucanía, Chile. *Compos. Paisaje Como Determinante Divers. Grup. Funcionales Aliment. Macroinvertebrados Acuát.* En *Ríos Araucanía Chile* 43, 186–200.
- Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., and Holling, C.S. (2004). Regime Shifts, Resilience, and Biodiversity in Ecosystem Management. *Annu. Rev. Ecol. Evol. Syst.* 35, 557–581.
- Friberg, N., Kronvang, B., Ole Hansen, H., and Svendsen, L.M. (1998). Long-term, habitat-specific response of a macroinvertebrate community to river restoration. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 8, 87–99.
- Goral, F., and Schellenberg, J. (2018). goeveg: Functions for Community Data and Ordinations.
- Gutiérrez-Cánovas, C., Sánchez-Fernández, D., Velasco, J., Millán, A., and Bonada, N. (2015). Similarity in the difference: changes in community functional features along natural and anthropogenic stress gradients. *Ecology* 96, 2458–2466.
- Harding, J.S., Benfield, E.F., Bolstad, P.V., Helfman, G.S., and Jones, E.B.D. (1998). Stream biodiversity: The ghost of land use past. *Proc. Natl. Acad. Sci. U. S. A.* 95, 14843–14847.
- Hauer, F.R., and Lamberti, G.A. (2006). *Methods in Stream Ecology* (Academic Press).
- Hogsden, K.L., and Harding, J.S. (2014). Isotopic metrics as a tool for assessing the effects of mine pollution on stream food webs. *Ecol. Indic.* 36, 339–347.
- Hughes, A.R., Byrnes, J.E., Kimbro, D.L., and Stachowicz, J.J. (2007). Reciprocal relationships and potential feedbacks between biodiversity and disturbance. *Ecol. Lett.* 10, 849–864.
- Jones, H.P., Jones, P.C., Barbier, E.B., Blackburn, R.C., Benayas, J.M.R., Holl, K.D., McCrackin, M., Meli, P., Montoya, D., and Mateos, D.M. (2018). Restoration and repair of Earth's damaged ecosystems. *Proc R Soc B* 285, 20172577.

- Kauffman, J.B., Beschta, R.L., Otting, N., and Lytjen, D. (1997). An Ecological Perspective of Riparian and Stream Restoration in the Western United States. *Fisheries* 22, 12–24.
- Lake, P.S. (2000). Disturbance, patchiness, and diversity in streams. *J. North Am. Benthol. Soc.* 19, 573–592.
- Lake, P.S. (2003). Ecological effects of perturbation by drought in flowing waters. *Freshw. Biol.* 48, 1161–1172.
- Legendre, P., and Legendre, L. (2012). Complex ecological data sets. In *Developments in Environmental Modelling*, (Elsevier), pp. 1–57.
- Maloney, K.O., and Weller, D.E. (2011). Anthropogenic disturbance and streams: land use and land-use change affect stream ecosystems via multiple pathways. *Freshw. Biol.* 56, 611–626.
- Mateos-Molina, D., Schärer-Umpierre, M.T., Appeldoorn, R.S., and García-Charton, J.A. (2014). Measuring the effectiveness of a Caribbean oceanic island no-take zone with an asymmetrical BACI approach. *Fish. Res.* 150, 1–10.
- Maul, J.D., Farris, J.L., Milam, C.D., Cooper, C.M., Testa III, S., and Feldman, D.L. (2004). The influence of stream habitat and water quality on macroinvertebrate communities in degraded streams of northwest Mississippi. *Hydrobiologia* 518, 79–94.
- McCabe, D.J., and Gotelli, N.J. (2000). Effects of disturbance frequency, intensity, and area on assemblages of stream macroinvertebrates. *Oecologia* 124, 270–279.
- McCain, M., Fuller, D., Decker, L., and Overton, K. (1990). Stream habitat classification and inventory procedures for northern California. USDA For. Serv. Pac. Southwest Reg. FHR Curr. San Franc. CA 94111 1–16.
- Merritt, R.W., Cummins, K.W., and Berg, M.B. (2008). *An Introduction to the Aquatic Insects of North America* (Kendall/Hunt Publishing Company).
- Mueller, M., Pander, J., and Geist, J. (2013). Taxonomic sufficiency in freshwater ecosystems: effects of taxonomic resolution, functional traits, and data transformation. *Freshw. Sci.* 32, 762–778.
- Mykrä, H., and Heino, J. (2017). Decreased habitat specialization in macroinvertebrate assemblages in anthropogenically disturbed streams. *Ecol. Complex.* 31, 181–188.
- Newbold, T., Hudson, L.N., Hill, S.L.L., Contu, S., Lysenko, I., Senior, R.A., Börger, L., Bennett, D.J., Choimes, A., Collen, B., et al. (2015). Global effects of land use on local terrestrial biodiversity. *Nature* 520, 45–50.

- Nimmo, D.G., Mac Nally, R., Cunningham, S.C., Haslem, A., and Bennett, A.F. (2015). Vive la résistance: reviving resistance for 21st century conservation. *Trends Ecol. Evol.* *30*, 516–523.
- Noss, R.F. (1990). Indicators for Monitoring Biodiversity: A Hierarchical Approach. *Conserv. Biol.* *4*, 355–364.
- Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., et al. (2018). *vegan: Community Ecology Package*.
- Palmer, M.A., Hondula, K.L., and Koch, B.J. (2014). Ecological Restoration of Streams and Rivers: Shifting Strategies and Shifting Goals. *Annu. Rev. Ecol. Evol. Syst.* *45*, 247–269.
- R Core Team (2018). *R: A Language and Environment for Statistical Computing* (Vienna, Austria: R Foundation for Statistical Computing).
- Roedenbeck, I., Fahrig, L., Findlay, C.S., Houlihan, J., Jaeger, J., Klar, N., Kramer-Schadt, S., and van der Grift, E. (2007). The Rauischholzhausen Agenda for Road Ecology. *Ecol. Soc.* *12*.
- Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., et al. (2000). Biodiversity - Global biodiversity scenarios for the year 2100. *Science* *287*, 1770–1774.
- Stanley, E.H., Powers, S.M., and Lottig, N.R. (2010). The evolving legacy of disturbance in stream ecology: concepts, contributions, and coming challenges. *J. North Am. Benthol. Soc.* *29*, 67–83.
- Thorp, J.H. (1991). *Ecology and Classification of North American Freshwater Invertebrates* (Academic Press).
- Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., and Cushing, C.E. (1980). The River Continuum Concept. *Can. J. Fish. Aquat. Sci.* *37*, 130–137.
- Verdonschot, P.F.M. (2009). Impact of Hydromorphology and Spatial Scale on Macroinvertebrate Assemblage Composition in Streams. *Integr. Environ. Assess. Manag.* *5*, 97–109.
- Violin, C.R., Cada, P., Sudduth, E.B., Hassett, B.A., Penrose, D.L., and Bernhardt, E.S. (2011). Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. *Ecol. Appl.* *21*, 1932–1949.

Wohl, E., Angermeier, P.L., Bledsoe, B., Kondolf, G.M., MacDonnell, L., Merritt, D.M., Palmer, M.A., Poff, N.L., and Tarboton, D. (2005). River restoration. *Water Resour. Res.* *41*, W10301.

CHAPTER 4

LIMITED NEAR-TERM EFFECTS OF CRANBERRY BOG RESTORATION ON IN-STREAM ECOSYSTEM STRUCTURE AND FUNCTION

ABSTRACT

Ecosystems are shaped by the interactions of communities with their environment. This balance can be shifted in multiple directions, with communities altering their habitats to their benefit or detriment, or changes to the abiotic conditions from external forces applying selective filters against community members. In the context of restoration, where the environment is targeted through directed disturbance to create specific alterations to system processes, these ecosystem functions must be evaluated to determine if changes have occurred in the near-term or are following a trend of longer changes over decades of on-site change. In this study, I used a Before-After-Control-Impact (BACI) design to investigate the effects of disturbance regimes on ecosystem level functional diversity in a restored flow-through cranberry bog in Plymouth, MA. Two control treatments were utilized to account for regional change through time, a least-impacted stream system (i.e. never used for agriculture) and an active flow-through cranberry bog. Seasonal water column chemistry measurements were coupled with annual (Summer) stream metabolism, nutrient limitation, and stable isotope derived functional food web measures for three years, beginning one year prior to the restoration in 2015. In contrast to expectations, few significant changes were observed in any functional measure except for stream metabolism,

which demonstrated pulse disturbance patterns of rapid change and recovery. Thus, we see that functional change in the near-term for this restoration project are limited in scope and demonstrate largely the impact of restoration's perturbation activities.

INTRODUCTION

Ecosystems are comprised of both abiotic factors and the communities that they contain. As such, ecosystems are inherently dynamic, with structures and processes driven by changes to the physical environment as well as community shifts (Folke et al., 2004). Many models of ecosystem community succession depend on predictable changes that are either allogenic, i.e. driven by abiotic factors, or autogenic, i.e. driven by intra- and interspecific competition and migration (Folke et al., 2004). In systems with less predictable drivers, a more tractable framework for considering community composition is one of multiple stable states, established by different pressures and attractors (Holling, 1996). Beisner et al. (2003) propose a method for visualizing this potential landscape as a ball and cup model, where the landscape's terrain marks changes to community dynamics including birth and death rates, while the ball on its surface represents the current community composition. Thus, the ball is drawn into basins of stability defined by changes to the community parameters, or momentum it accumulates through shifts in its abundances or assemblage species.

This paradigm of ecosystem community is defined solely by changes to the community dynamics or composition. However, changes to community rates are often controlled by abiotic processes. For example, invertebrate diversity and abundances can be

controlled by flow conditions (Bunn and Arthington, 2002), while nutrient availability limits plant growth (Newbold et al., 1982). Indeed, many documented state shifts are measured through community assemblage changes, but are attributable to abiotic shifts. Eutrophication has been tied to altered communities (Villéger et al., 2010) and sedimentation increases have been shown to potentially lead to trout extirpation (Harvey et al., 2009). However, not all state shifts are dependent on shifts to the physical or chemical environment. Many studies of invasive species have shown community shifts, occurring without change to the abiotic environment (Stotz et al., 2017). Thus, while pursuing the possibility of ecosystem state shift, abiotic change as well as community composition change are both potential leading indicators of movement.

Assessing community state shifts is inherently a difficult process, as changes to community composition or the physical habitat do not happen in a vacuum. Communities may experience some level of stochasticity without meaningfully shifting in their relative state, and determination of a given system's resistance or resilience often requires long term data on annual fluctuations and background levels of disturbance (Carpenter and Brock, 2006). Even then, it may take years for the community to respond in a manner that is detectable, even if the triggering event can be reasonably assumed (Hewitt and Thrush, 2010). However, abiotic indicators may exhibit increased variability around distinguishing events and may act as leading indicators of an eventual state shift (Carpenter and Brock, 2006). Examining ecosystem functionality thus may provide some important baseline data as well as potential first evidence of eventual community shift. The ability to distinguish drivers from secondary effects also requires a strong baseline of data regarding the community and its physical and chemical habitat.

The principles behind alternative community states created through alterations to the abiotic environment underpin an essential part of ecological restoration. Ecological restoration applies the concepts of disturbance ecology to attempt to shift the community from one impaired state to an ameliorated through the use of directed perturbations, termed active restoration (Stanley et al., 2010; Wohl et al., 2005). This sits in contrast with the historical use of passive restoration, where natural succession and background environmental variation would be enough to alter a system towards a target state (Kauffman et al., 1997). When active restoration is used to restore loss of function at the ecosystem level, including nutrient and sediment fluxes, detrital input, and other features, the end result is process-based restoration (Wohl et al., 2005). It is precisely these restoration projects that prompt the question of whether the engineered disturbances do alter ecosystem processes and eventually result in a shift in the community.

STUDY AIM AND HYPOTHESES

The goal of this study was to determine if process-based restoration resulted in measurable shifts to the ecosystem processes or functional attributes of the community. Given the relative importance in aquatic systems of water as a medium for material flux, source of dissolved oxygen, and its role as fundamental habitat, I investigated several qualities to assess function in these roles. Specifically, I chose to analyze water column chemistry, patterns of dissolved oxygen demand and supply, and whether algal growth was limited by nutrient availability. These parameters were monitored and assessed for changes before and after restoration. Additionally, to examine functional changes to the community

that might precede or coincide with compositional shifts, functional food web analysis was conducted using stable isotopes of carbon and nitrogen to observe any changes to carbon sources, trophic length, or niche similarity within the fish and macroinvertebrate assemblages. To achieve the goal of detecting abiotic shifts and functional community change, I explored 4 primary hypotheses.

Firstly, I hypothesized that water column chemistry, an ecosystem compositional attribute, would undergo a significant and detectable shift post-restoration that would be maintained through the course of the study. As water column chemistry is widely regarded as a snapshot of current conditions, rather than an integrative one (Hauer and Lamberti, 2006), I predicted that this would reflect the most immediate alteration to the system following restoration activities. Additionally, the restoration's design to maximize exposure to previously buried peat sediments should result in detectable changes to dissolved and particulate nutrients as well as suspended solids within the system.

Secondly, I hypothesized that diurnal changes in dissolved oxygen fluctuation, as measured by whole stream metabolism analysis, an ecosystem functional attribute, would not significantly shift over the course of the study. The incorporation of dissolved oxygen in streams is driven in large part by physical reaeration and the photosynthetic addition of oxygen by algae and macrophytes, while decreases are driven by biological oxygen demand of the animal and microbial community (Bales and Nardi, 2007). As the direct effects of the restoration do not provide for significant reaeration, and macrophytes would likely only be limited by decreased solar input (Vannote et al., 1980), I did not anticipate any shifts in dissolved oxygen patterning during the course of the study, as increased canopy cover would be necessary to alter macrophyte abundance in a significant way.

However, reconnecting the stream bed with buried peat sediments and disrupting existing macrophyte growth could generate a decrease in gross primary production as well as an increase in biological oxygen demand following restoration, particularly in the first year post-restoration.

Third, I hypothesize that food web dynamics, ecosystem structural attributes, will show reductions in trophic length and carbon source diversity in the first year post-restoration due to channel reconstruction activities. These reductions will be seen through decreases in ranges for nitrogen and carbon stable isotope ratios obtained from the fish and macroinvertebrate assemblages, reflecting disturbance-based reductions to food web parameters. I predict that the second year post-restoration will show a widening of all food web metrics, as new primary producers establish, potential new predators enter, and the system provides stability for increased trophic redundancy. These will be reflected in increases in carbon range, nitrogen range, and a decrease in nearest neighbor distance, respectively.

Finally, I hypothesized that experiments testing nutrients limitation, an ecosystem functional attribute, would show the development of phosphate limitations within the system following restoration. Preliminary research at CBR demonstrated no significant nitrogen or phosphorous limitations to algal growth (Franck, 2017), likely a consequence of nutrient additions during decades of farming and legacy fertilizers within the system soil. With the dredging of entirely new stream channel along the majority of the system, with the stream bottom below the depth of agricultural topsoil, I predicted the legacy effects of fertilizer leaching would be diminished. Thus, as freshwater streams are typically

phosphorous limited (Hauer and Lamberti, 2006), I predicted that the final year of the study would show phosphorous limitation as a major control on algal growth.

MATERIALS AND METHODS

Study Area

I conducted this research at three (3) treatments, representing directed pulse disturbance (CBR, Tidmarsh Farms), background disturbance (LIR, Mashpee River), and an agricultural press disturbance (ACB, Active cranberry bog site) in three separate low-order (1st-3rd) streams fed by primarily groundwater and drainage fed ponds in the Atlantic Coastal Pine Barrens Ecoregion of Massachusetts (Figure 1). CBR was taken out of production in 2010, thus entering a phase of passive restoration (Kauffman et al., 1997). CBR then underwent active restoration in Winter 2015/2016. Each treatment consists of three (3) replicate stations defined as 100m length segments where data was collected. Each station was separated by at least 100m in river channel length and sampled seasonally for water chemistry, and annually for all other data.

Seasonal Water Chemistry

Water column chemistry was collected and analyzed following standard methods for surface water grab samples (Hauer and Lamberti, 2006). Briefly, water samples were taken from surface waters in an acid washed 4L Nalgene cubitainer. Samples were stored on ice for transport back to the laboratory, held at 4 °C and processed within 48 hours. Water was filtered on 1µm A/E glass fiber filters and processed for total suspended solids, chlorophyll a content (Turner 10AU Fluorometer), particulate carbon (Costech Elemental

Analyzer), particulate nitrogen (Costech Elemental Analyzer), and particulate phosphorous (Perkin Elmer UV-Vis Spectrophotometer) following standard protocols (Clesceri et al., 1998). Water filtrate was analyzed for total dissolved inorganic nutrients of ammonia (31-107-06-1-B), nitrate (31-107-01-1-E), and orthophosphate (31-115-01-1-I) via Flow Injection Analysis (HACH/Lachat QuikChem 8500). All analyses were run in duplicate to account for potential instrument error.

Duplicate samples were averaged and used in analysis as a single value. Water chemistry variables were grouped by sampling station and sampling event coding and analyzed using a principle component analysis in R (R Core Team, 2018). Analysis was conducted separately for each season due to the known variability in water column chemistry during the water year (Wold and Hershey, 1999).

Whole Stream Metabolism

Whole stream metabolism was determined annually in the summer by measuring diel dissolved oxygen dynamics following Hauer and Lamberti (2006). Dissolved oxygen (D.O.) and temperature were monitored by a submerged data logger (Onset U26-001 Dissolved Oxygen Logger) at 10 minute intervals both upstream and downstream of a 100m reach for a minimum of 30 hours, including two over-night periods (Figure 2). Due to time and material constraints, only 2 stations per treatment were utilized each year, located at the upstream most and downstream most stations to provide the maximum range of variability. This subsequently reduces power to detect changes through time and between treatments. Stream physical parameters such as flow, depth and width were concurrently recorded such that water travel time and D.O. reaeration across the reach could be calculated. Briefly, flow was recorded using a Flo-mate 2000 portable flow meter

(Marsh-McBirney) at both the 20% and 80% depth across midpoint transects at each macrohabitat within the 100m reach. A minimum of 3 transects were performed, following the protocol laid out in the Basin Area Stream Survey (McCain et al., 1990). Flow, depth, and width for each transect were combined as a weighted average based on macrohabitat length and used to calculate travel time over 100m and the surface renewal model (SRM) derived reaeration coefficient (Hauer and Lamberti, 2006). The resultant information along with the D.O. and temperature time series were used to calculate gross primary production (GPP), net daily metabolism (NDM), and production to respiration ratio (P/R ratio) based on changes to D.O. over the photoperiod and overnight. Calculations followed Hauer and Lamberti (2006) single station methods, and utilized functions from the StreamMetabolism package in R (Sefick, Jr., 2016).

These ecosystem function values were independently calculated on both the upstream and downstream dissolved oxygen logs for each station and were then treated as analytical replicates. All three parameters in each year/treatment pairing were analyzed following effect size for a BACI design:

$$d = (A_I - A_C) - (B_I - B_C)$$

Where A represents the mean parameter value in the After period while B represents the mean value for the parameter Before (i.e. in 2015, the initial year of study) for both I , impact treatment, and C , control treatment (Christie et al., 2019).

Confidence intervals (95%) were generated based on these effect sizes and pooled variance:

$$s_p^2 = \frac{(n_{BI} - 1) * s_{BI}^2 + (n_{BC} - 1) * s_{BC}^2 + (n_{AI} - 1) * s_{AI}^2 + (n_{AC} - 1) * s_{AC}^2}{(n_{BI} + n_{BC} + n_{AI} + n_{AC} - 4)}$$

$$CI = d \pm 1.96 * \sqrt{\frac{s_p^2}{n_{BI}} + \frac{s_p^2}{n_{BC}} + \frac{s_p^2}{n_{AI}} + \frac{s_p^2}{n_{AC}}}$$

Where s^2 and n represent the variances and sample size for each set of treatment/period combinations (Christie et al., 2019). This calculation was done for both 2016 and 2017 as separate After periods, as there is no assumption of a stepwise change immediately following restoration combination, comparable to a Before-During-After-Control-Impact study (Roedenbeck et al., 2007). Additionally, due to the differences between control treatments, separate analyses were run treating ACB or LIR as the control treatment. Effect sizes were determined to be statistically significant if the confidence interval did not overlap zero.

In order to assess diversity changes at CBR relative to LIR and ACB conditions, Welch's t-tests were computed pairwise within years, comparing CBR to ACB and CBR to LIR, to determine if CBR's richness or Shannon's diversity altered relative to either reference condition. To increase power without increasing Type I error, pairwise comparisons between LIR and ACB were not conducted. Significance was adjusted due to multiple comparisons using Bonferroni's correction ($\alpha=0.008$, $n=6$) (Legendre and Legendre, 2012).

Isotopic Food Web Analysis

Food web functional structure was determined by carbon and nitrogen stable isotope analysis of organisms from the fish and FMI communities. The protocol follows a modified form from Hogsden and Harding (2014). Briefly, fish vouchers, collected during summer assemblage surveys as described below, had dorsal muscle tissue extracted. Freshwater macroinvertebrates (FMI) were collected in the summer, separately from seasonal collections, field sorted to the lowest possible taxonomic level, and frozen. Due to low dry mass, pooling of small organisms was required to create a detectable signal. To avoid bias between pooled and non-pooled samples, all samples regardless of tissue quantity were pooled to include at least 3 individuals where possible. All specimens were dried at 60°C, homogenized to a fine powder using a Qiagen TissueLyzer and stainless-steel beads, and analyzed by Isotopic-Ratio Mass Spectrometry (IRMS). Stable carbon and nitrogen analyses were conducted at the University of Massachusetts Boston's Environmental Analytical Facility using a Delta V Isotope Ratio Mass Spectrometer (IRMS, Thermo-Fisher), with samples being combusted with an Elemental Analyzer 4100 (Costech). Isotope ratios were expressed in the $\delta^{13}\text{C}$ or $\delta^{15}\text{N}$ notation = $(R_{\text{sample}}/R_{\text{standard}}) - 1000$, where $R = {}^{13}\text{C}/{}^{12}\text{C}$ or ${}^{15}\text{N}/{}^{14}\text{N}$ using international standards: VPDB for carbon and air for nitrogen. Data are expressed in parts per thousand (‰).

Fish were sampled annually during the summer using double-pass electroshock surveys (Kimmel and Argent, 2006) following IACUC protocol (IACUC2013005). In brief, block-nets were deployed across the width of the stream at the upstream and downstream end of the station and was secured at both banks and on the stream bottom. The survey team consisted of at least 3 members, 2 netters/bucketers and one “operator”

wearing a Smith-Root backpack electrofisher (LR-24; Smith-Root, Inc.). The electrofisher was used to create an electric current along an ellipse, temporarily paralyzing fish for collection. Amperage and voltage of the electrofisher was set based on conductivity of the water in accordance with manufacturer recommendations. Paralyzed fish were netted and deposited into aerated buckets filled with stream water. Collection occurred downstream to upstream, with the operator moving in a serpentine fashion to create an impassible electrical field across the stream width. Collected fish were identified to the species level and a maximum of 3 voucher organisms per species were retained for isotopic analysis – all other fish were returned to the stream downstream of the site after each pass. Sediment was allowed to settle before the second pass. Fish assemblage data were analyzed as part of a larger study examining the impacts of restoration (see Dimino, in prep). Voucher specimens were stored on ice and frozen upon return to laboratory, as identification vouchers and for stable isotope analysis, as described above.

FMI were collected by multi-habitat dip-netting, following EPA protocols (Barbour et al., 1999). Briefly, in-stream macrohabitats were determined to the nearest 5% of stream area as determined via BASS assessment. Each 5% increment of habitat constitutes 1 collection point, or “jab,” which occurs in a 0.5 square meter area of the macrohabitat. Kicking or jostling with handheld net disturbs sediments, bank walls, aquatic vegetation, or root masses and a 500µm mesh D-shaped net was then swept 5-10 times through the water column, collecting suspended sediments and FMI. All 20 jabs were distributed as evenly as possible across the 100m station. The composite FMI collection is then transferred into a plastic white sorting tray sorted on site. Organisms were sorted and identified to the lowest possible taxonomic resolution (Order/Family) and up to 5

individuals per taxonomic group were removed from the tray and placed into glass vials. Sample organisms were then stored on ice and frozen upon return to the laboratory and processed as stable isotope samples as described above.

Resultant FMI and fish assemblage stable isotope values were used to compute a functional food web for each station (Figure 3), including total nitrogen delta range, total carbon delta range, and nearest neighbor distance averages and standard deviations (Hogsden and Harding, 2014; Layman et al., 2007). These values were then analyzed for each Treatment/Year combination using BACI effect size and pairwise within-year treatment comparisons as above for Whole Stream Metabolism.

Nutrient Limitation Experiments

Nutrient limitation experiments were performed annually following Gibeau and Miller (1989). Plastic vials were filled with nutrient diffusing agar gel, containing either 1M soluble ammonia and nitrate (Ammonium Nitrate, 2M total N), 1M soluble phosphate (Potassium Phosphate Monobasic), both, or no additional nutrients and capped with ceramic filters. Filled and capped vials were secured to racks and placed at the streambed of each station in areas of maximum solar exposure at each station and allowed to incubate for 3 weeks during the summer to allow for optimal algal growth. Each rack of vials was removed from the water column and individual vials were removed from the rack, frozen, until ready to analyze. In the lab, the porcelain caps were removed from the top of the vial, placed in buffered acetone for extraction and analyzed for chlorophyll *a* content following the same method described above for water chemistry, adjusting for surface area of filter rather than volume of filtrate. Limiting nutrient within each disturbance treatment and year was determined via ANOVA where groupings were given by experimental addition (i.e.

Control, Nitrogen, Phosphorous, and Nitrogen+Phosphorous) followed by Tukey's Honest Significant Difference (HSD) as a post-hoc procedure if significant. Tukey's HSD was chosen over Welsh's corrected t-tests due to the fact that the ANOVAs were within year and treatment analyses, which better supported an assumption of homogenous variance.

RESULTS

Seasonal Water Chemistry PCAs

Water chemistry values were expected to show initial differences between all three treatments, regardless of season, as a result of the particular land use associated with each treatment. I also expected water chemistry at CBR to change from 2015 to 2016 in response to the perturbation of the restoration and for that change to be maintained in 2017, a result of the relatively immediate shifts in groundwater and riparian zone inputs to the water column.

All 3 PCAs explain a large amount of the variance in water chemistry parameters, with variance explained ranging from 64.96% to 71.45% within the first two principle components (Table 1). The PCAs also appear to be largely driven by the dissolved water chemistry, with higher loading values for dissolved phosphate, ammonia, and nitrate than any other analyte regardless of season or principle component (Table 1).

Differences between treatments overall was greatest in Spring and Fall, and weakest during the Summer season (Figure 4). Although no additional analyses were performed on these analyses, a number of features can be visualized. While LIR is consistently grouped tightly together regardless of year or season, ACB exhibits grouping

of the 2016 and 2017 years separate from 2015 for both the Spring and Fall data (Figure 4). The change over time in CBR water chemistry likewise appears seasonally dependent. The PCA of Spring water chemistry demonstrates an initial change at CBR in 2016, however this is reversed in 2017 (Figure 4A). In the Summer, however, CBR 2016 and 2017 both appear different from the initial data collected in 2015 (Figure 4B). Finally, in the Fall, high variability in 2015 and 2016 make it difficult to determine if meaningful change has occurred in the water chemistry parameters (Figure 4C).

Whole Stream Metabolism

Diurnal fluxes in dissolved oxygen were not expected to change significantly during this study, except for potential disturbance effects in the initial year following restoration. I expected GPP, NDM, and P/R ratios to be unchanged relative to the initial conditions due to the lack of increased reaeration potential and the continued lack of canopy cover which would limit in-stream production. However, across time CBR experienced changes with large effect sizes for all 3 parameters in 2016 and lowered or opposite effects in 2017 relative to 2015, as measured relative to control treatments. This is consistent with the pulse effect of the restoration activities, an initial large change followed by correction. When comparing between treatments, significant differences were not consistent across years, suggesting high yearly variability in these ecosystem processes. It is also important to note that due to probe malfunction, 2017 ACB was limited to a single station and was thus excluded from pairwise statistical analysis.

For GPP, although large changes in effect size were seen (2016: $d_{LIR} = 7.960$ CI: [-6.457, 22.376]; $d_{ACB} = 4.879$, CI: [-2.457, 12.215], 2017: $d_{LIR} = 4.463$ CI: [-9.49, 18.415]; $d_{ACB} = -6.959$, CI: [-14.74, 0.821]), no significant difference was observed through time at

CBR (Figure 5A). Between treatments within years, there were no significant differences between CBR and either LIR or ACB (Welsh's t-tests, Appendix A.2) (Figure 5B).

Net daily metabolism showed significant decline through time at CBR between 2015 and 2016 relative to ACB ($d_{ACB} = -3.339$, CI: $[-6.296, -0.382]$) but not LIR ($d_{LIR} = -36.039$, CI: $[-106.326, 34.248]$) and there was no significant change in 2017, although the effect was in the opposite direction ($d_{LIR} = 13.488$, CI: $[-63.353, 90.328]$; $d_{ACB} = 4.214$, CI: $[-10.162, 18.589]$) (Figure 6A). Between treatments, there were no significant differences between CBR and either other treatment (Welsh's t-tests, Appendix A.2) (Figure 6B).

Finally, the P/R ratio was significantly different from 2015 to 2016 for CBR relative to LIR ($d_{LIR} = 0.347$, CI: $[0.295, 0.399]$), but not ACB ($d_{ACB} = -0.040$, CI: $[-0.559, 0.478]$) (Figure 7A). There was no significant change relative to control in 2017 ($d_{LIR} = -0.088$, CI: $[-0.572, 0.395]$; $d_{ACB} = -0.543$, CI: $[-1.369, 0.283]$). Between treatments, CBR differed from LIR in 2016 (Welsh's $t=1107.659$, $v=1.001$, $p<0.001$) (Figure 7B) but not in 2017.

Isotopic Food Web Analysis

Functional food web dynamics were expected to change following restoration, with an initial decline in trophic area, and carbon and nitrogen range as disturbance limits the food web, followed by an expansion of food web metrics, including increased trophic area, carbon and nitrogen ranges, and a decrease in nearest neighbor distance. However, across time, CBR experienced significant effect sizes only in comparison to LIR for trophic area and nitrogen range. When comparing between treatments, only in 2017 were significant differences in trophic metrics observed, between CBR and LIR for carbon range, however due to low sample size in ACB for 2017, direct comparisons were unable to be performed.

Trophic area at CBR declined slightly in 2016 and more severely in 2017, although significant change only occurred in 2017 relative to LIR ($d_{LIR} = -78.292$, CI: [-126.272, -30.311]; $d_{ACB} = -13.521$, CI: [-64.882, 37.840]) (Figure 8A). This however may be largely due to an increase in trophic area of LIR in 2017. Between treatments, there were no significant differences between CBR and either LIR or ACB (Welsh's t-tests, Appendix B.2) (Figure 8B).

Nitrogen range at CBR exhibited more minor shifts in effect sizes, and significant change was observed only in 2017 relative to LIR ($d_{LIR} = -11.124$, CI: [-18.664, -3.585]; $d_{ACB} = -2.049$, CI: [-12.649, 8.551]) (Figure 9A). Between treatments, no significant differences between CBR and either of the 2 control treatments was observed (Welsh's t-tests, Appendix B.2) (Figure 9B).

Carbon range at CBR did not experience a significant change from through time relative to either control (BACI effect size, Appendix B.1), (Figure 10A). Between treatments, only in 2017 was there a significant difference between CBR and LIR (Welsh's $t = -24.612$, $v = 2.025$, $p = 0.0015$) due to the increase in carbon range for LIR in 2017 (Figure 10B).

Nearest Neighbor Distance (NND), a measure of food web redundancy, did not exhibit any significant change across time for CBR relative to either LIR or ACB (BACI effect size, Appendix B.1) (Figure 11A). Although there was an increase in NND during 2016 for CBR, this increase reversed in 2017 and no changes were significant. Between treatments, there was no significant differences in any year (Welsh's t-tests, Appendix B.2) (Figure 11B).

Standard Deviation of Nearest Neighbor Distance (SDNND) followed a similar pattern to NDD (Figure 12), with no significant changes either across time or between CBR and other treatments (BACI effect size and Welch's t-tests, Appendix B.1 and B.2, respectively). CBR did exhibit an increase in 2016, but fell again in 2017 and neither change was significantly different from each other or 2015 (Figure 12A). Likewise, between CBR and other treatments within year, no significant differences were found (Figure 12B).

Nutrient Limitation Experiments

Phosphorous limitation was expected to develop at CBR in response to the restoration event, as freshwater systems are typically phosphorous limited and remediation of legacy sediments was expected to result in algal control by phosphate availability. However, no nutrients were determined to be limiting at any of the sites across the years of study.

The only significant difference in any year/treatment pairing was for ACB in 2015 and 2016, ($F=6.377$, $df=3$, $p=0.016$ and $F=23.50$, $df=3$, $p=2.54e-4$ respectively), with both having maximal chlorophyll growth for the no nutrient addition (i.e. Control) (Figure 13).

DISCUSSION

Overall, the results of my study lead to four main findings. First, water column chemistry is highly seasonal, but the primary drivers of differences between years and treatments is dissolved inorganic nutrients. Second, in all 3 parameters for whole stream metabolism, CBR exhibited large effect size swings, increasing all three parameters in

2016 and rebounding in 2017, suggesting disturbance impacts from the restoration that were not maintained into the second year post-restoration. Third, food web dynamics showed high variability through time for all 3 treatments, regardless of parameter investigated, with CBR showing no maintained changes or higher variability than either alternative disturbance regime treatment. Finally, neither dissolved nitrogen nor dissolved phosphorous were determined to be limiting in any treatment over any year and no change was observable through time.

Seasonal Water Chemistry

Although there is considerable variation in water chemistry parameters, including in relative relationships across seasons, there is still substantial information to be gleaned from these water chemistry “snapshots.” PCA data for the Spring season suggest a change in CBR water chemistry in 2016, making it different from both 2015 and 2017. As Spring 2016 was the nearest timepoint to the active phase of restoration, this shift may be a response to the disturbance features of the restoration’s rechannelization and mobilization of sediments and peat derived nutrients, as is seen in remediation efforts (Barton, 1977). However, this pattern is not equally observed in the Summer and Fall data, suggesting either the effect was transitory or confounding of time and season effects. Across all ordinations, dissolved nutrients (i.e. Nitrates, Orthophosphate, and Ammonia) are strongly associated with the first two principle components. Much of the variation observed across all three treatments can thus be tied to dissolved nutrients, which are more closely tied to agricultural and sediment input from the watershed (Gabriele et al., 2013). Thus, restoration’s ability to alter water chemistry may be limited without controls on upstream watershed fluxes of nutrient loaded soils more broadly in Southeastern Massachusetts.

Whole Stream Metabolism

As measured by effect size, CBR exhibited a strong increase in GPP and P/R ratio, alongside a decrease in net daily metabolism between 2015 and 2016. This contrasts with my expectation that oxygen dynamics would not change over the course of the study. The first year post-restoration increase in primary production suggests that submerged aquatic vegetation growth benefited from the restoration, possibly due to increased nutrient availability from the peat layer (Guecker et al., 2009; Izagirre et al., 2008) or due to the increase in rooting ability with reduction in sandy bottomed areas (Madsen and Adams, 1989).

Between treatments, the relative relationships for each of the parameters varied across years, largely due to the high variability between sites within a treatment/year. This suggests that whole stream metabolism dynamics are sensitive to both spatial and temporal changes within a riverine system. As a result, the early dynamics of production and respiration following restoration should not be assumed to be typical of the post-restoration status and merit continued monitoring in the future, especially as reforestation affects in-stream production through light shading and temperature reduction (Sudduth et al., 2011).

Isotopic Food Web Dynamics

The five measured parameters from stable isotope ratios of carbon and nitrogen within the fish and macroinvertebrate assemblage represent indices for food web width, length, and niche redundancy (Hogsden and Harding, 2014; Layman et al., 2007). Ultimately, significant changes were limited to trophic area and nitrogen range in 2017, and only relative to LIR, which itself exhibited a large increase in 2017. This is in contrast to my expectation that disturbance would shorten the trophic length (Power et al., 1995),

and reduce specialist niches (Angermeier, 1995). Although not significant, there is an increase in nearest neighbor distances and standard deviation of nearest neighbor distances at CBR in 2016, suggesting there is greater separation of each individual organism from each other's food web niche (Layman et al., 2007). As functional changes are thought to follow from compositional ones rather than the reverse (Palmer et al., 2014), continued investigation of the CBR food web in the future may provide more substantive changes, particularly following canopy recovery.

Nutrient Limitation

Pristine freshwater systems are typically characterized by phosphorous limitation, although they may alternatively be limited by nitrogen or both nutrients (Elser et al., 2007; Hauer and Lamberti, 2006). As such, I expected that following restoration CBR would begin to show signs of nutrient limitation. However, the lack of nutrient limitation at any of the treatments in any year suggest a more regional pattern of streams not limited by either nitrogen or phosphorous. One potential explanation is that all three treatments are coastal streams draining from upstream lakes. This introduces the potential for the introduction of phosphorous and nitrogen by outflowing eutrophic water (Mattson et al., 2004). Notably, variability both within and between nutrient group is highest in 2014 and 2015, potentially due to the use of additional analytical replicates per station (excluded from the analysis for consistency across time) in those years. The addition of extra diffusing gels may have created a gradient of each nutrient addition within the analytic replicates, possibly increasing their variance. Alternatively, this may simply represent increased technical skill in deployment and analysis.

CONCLUSIONS

Process based restoration has the stated goal of re-establishing key ecosystem interactions in impaired freshwater areas, with the assumption that once these functions are restored the biological community will follow (Poff, 1997; Wohl et al., 2005). Therefore, examination of these matter and energy transfers are key to identifying restoration's ability to create the template for target biological assemblages to return. While function is thought by some to follow from composition changes (Palmer et al., 2014), the early changes following restoration allow for evaluation of the creation of a new distinct environmental template, allowing for reorganization (Stanley et al., 2010).

Among the parameters examined in this study, we had expected a few distinct changes following an expectation of how the system would change following perturbation. Although water chemistry was expected to change rapidly and stabilize quickly, I found limited evidence for this immediate and maintained transition. Likewise, nutrient limitation experiments were expected to reflect the relatively rapid reduction in legacy fertilizer impacts, but no nutrient limitations were observed, suggesting either higher than expected legacy effects, or consequences of general eutrophication in the region's lakes.

I also saw neither initial reduction in food web metrics, nor significant change at CBR over the course of the study, which suggests that any compositional changes are not being accompanied by shifts in food web length, breadth, or redundancy. However, this is both potentially due to the lack of specialist species in the pre-restoration site which would be expected to be lost in the perturbation, combined with limited ability for rapid dispersal of higher trophic level organisms (i.e. fish) and the lack of significant change to carbon

input before reforestation. Isotope metrics are also sensitive to taxa sampled and need to be compared against those sampled more rigorously and laboratory identified.

Finally, on the other end of the spectrum, I expected no significant changes to occur in whole stream metabolism due to the expectation that this would only be substantially changed by canopy cover and reduction in macrophyte growth. However, production and metabolism show clear evidence of the effects of perturbation in 2016 that is abrogated in 2017, suggesting more evidence of a pulse disturbance.

Taken together, these parameters suggest that limited functional change has occurred, and where alterations are seen, they are best explained by the pulse disturbance effect of rechannelization and other restoration activities. This matches base expectations that functional change occurs over longer time scales, often with the reforestation of riparian vegetation after restoration (Palmer et al., 2014; Sudduth et al., 2011). However, investigation of functional change is limited in most restoration projects, making it a key feature for examination, especially in long-term studies regarding change relative to the age of restoration (Evju et al., 2020).

ACKNOWLEDGEMENTS

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TABLES

Table 1: Loading values for each original water chemistry analyte in the first two principle components after PCA analysis. Each analyte loading score is given for the individual seasonal PCA of water chemistry parameters.

Analyte	<u>Spring</u>		<u>Summer</u>		<u>Fall</u>	
	PC1 (40.80%)	PC2 (24.16%)	PC1 (48.62%)	PC2 (22.83%)	PC1 (50.67%)	PC2 (20.31%)
Dissolved Phosphate (PO ₄ ²⁻)	-0.4304	-0.428	-0.4962	0.7401	-0.2364	0.5978
Dissolved Ammonia (NH ₃)	0.1071	-0.6118	-0.1938	0.5616	0.0983	0.8812
Dissolved Nitrate (NO ₃ ⁻²)	-0.1518	-0.4971	0.0316	0.518	0.0093	0.7427
Total Suspended Solids	-0.3115	-0.1593	-0.279	0.2682	0.4004	-0.0027
Chlorophyll (a)	-0.305	0.2308	-0.2983	-0.2156	0.2091	-0.0829
Particulate Carbon	-0.3238	0.0036	-0.2295	-0.0613	0.1925	-0.0063
Particulate Nitrogen	-0.2193	0.1151	-0.1573	-0.071	0.1112	-0.0091
Particulate Phosphate	-0.0322	-0.0166	-0.0144	-0.001	0.0061	0.0013
Nitrogen/Carbon Ratio	-0.0237	0.1992	0.0231	-0.0878	-0.008	-0.0129

FIGURES

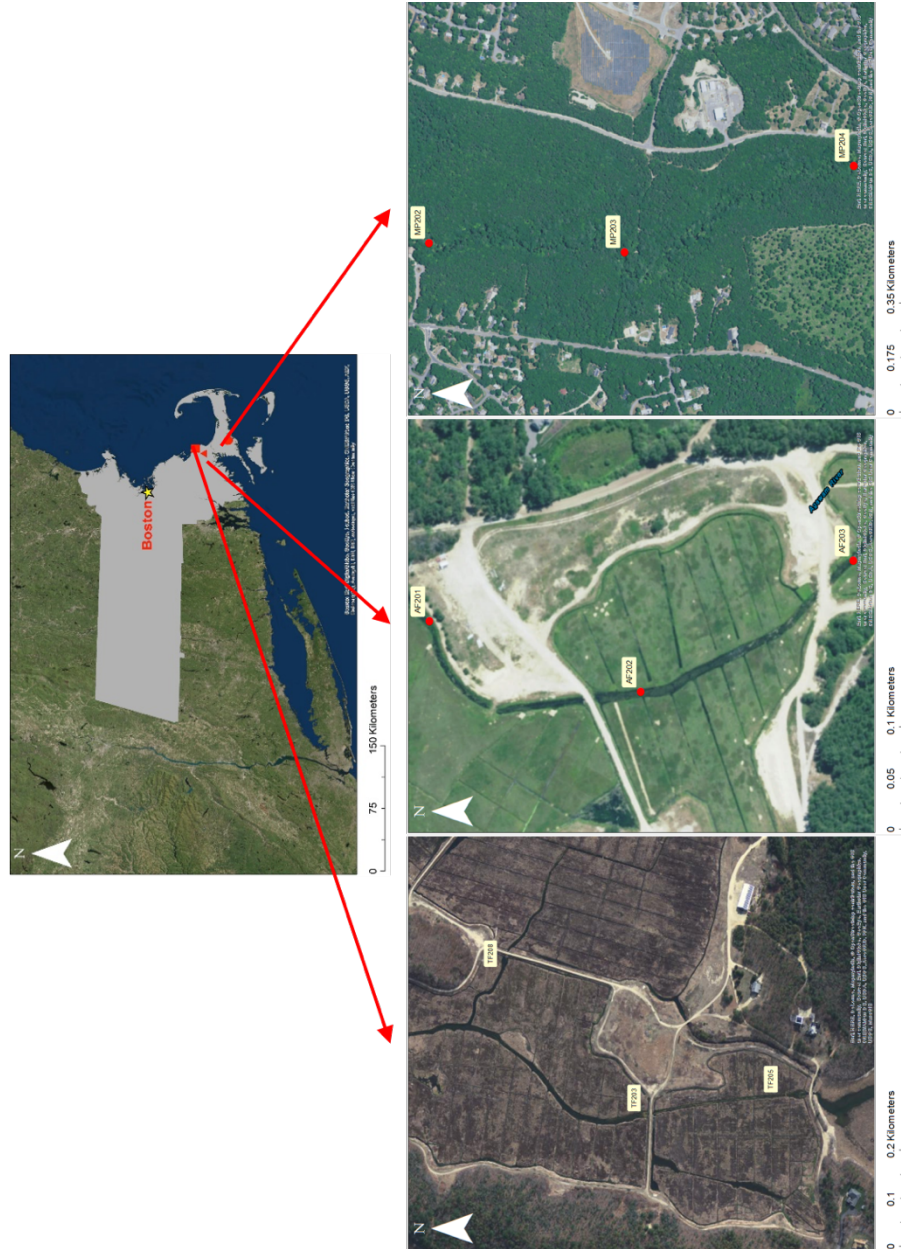


Figure 1: Experimental treatment study locations and their relative locations in Southeastern Massachusetts. From left to right: Tidmarsh Farms (CBR, pre-restoration imagery), an active flow-through cranberry bog (ACB), and the Mashpee River flowing through the Trustees of Reservations conservation land (LIR) (ESRI). Map made by S. McCanty.

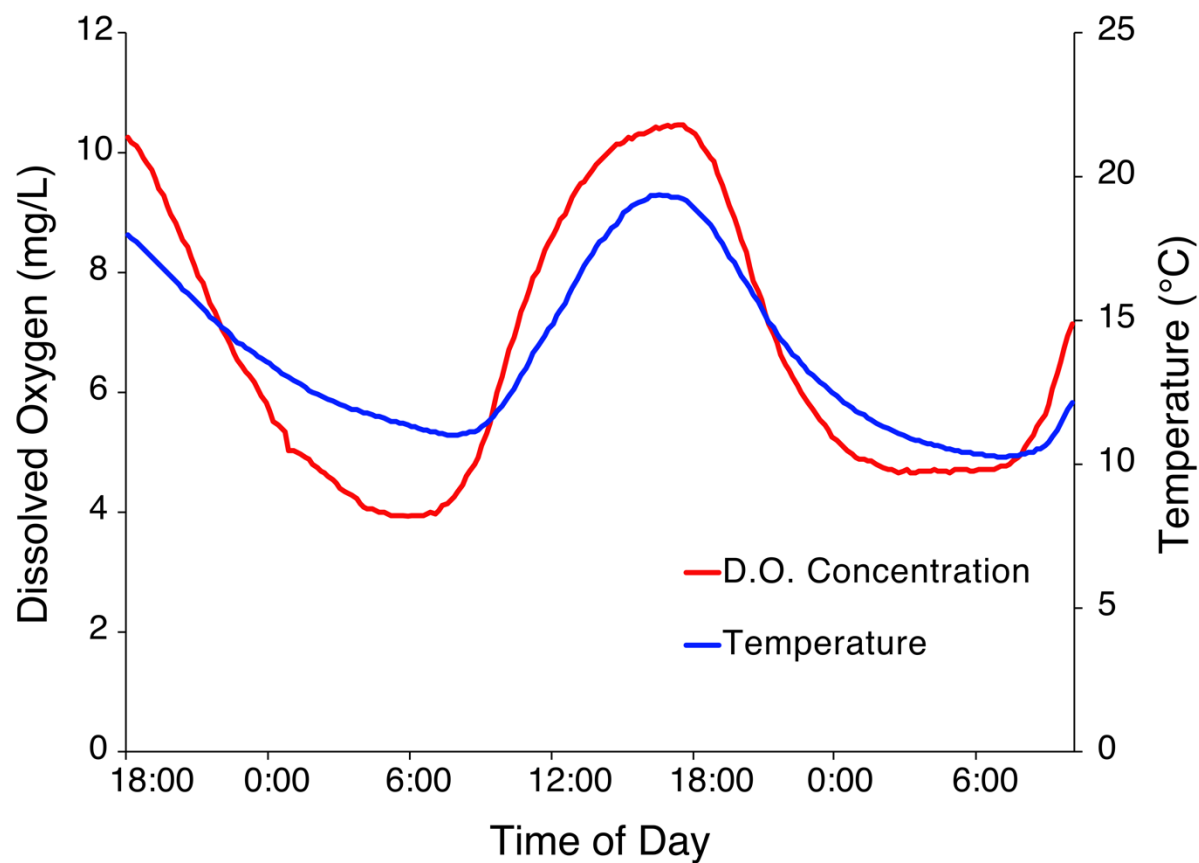


Figure 2: Example graph of dissolved oxygen and temperature logging over a 40-hour period at CBR station CBR205 in Summer 2015 (upstream location).

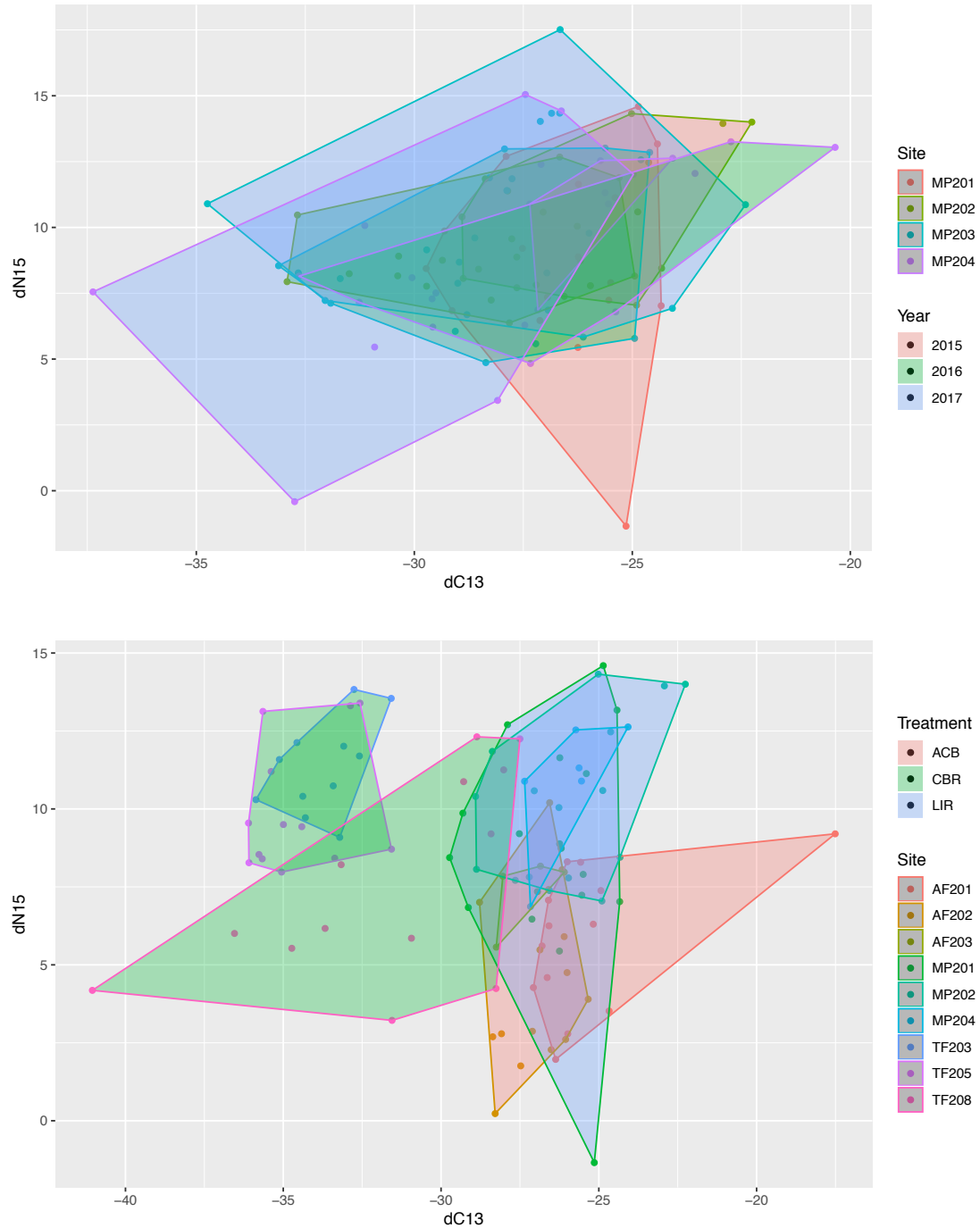


Figure 3: Example graphs of isotopic analysis for each food web. Graphs are broken down by Treatment (LIR, A) or Year (2015, B), with convex hulls capturing the extent of each station's trophic area. These polygons were used in the calculations of Trophic Area, Total Nitrogen Extent, Total Carbon Extent, Nearest Neighbor Distance, and Standard Deviation of Nearest Neighbor Distance (Figures 8-12).

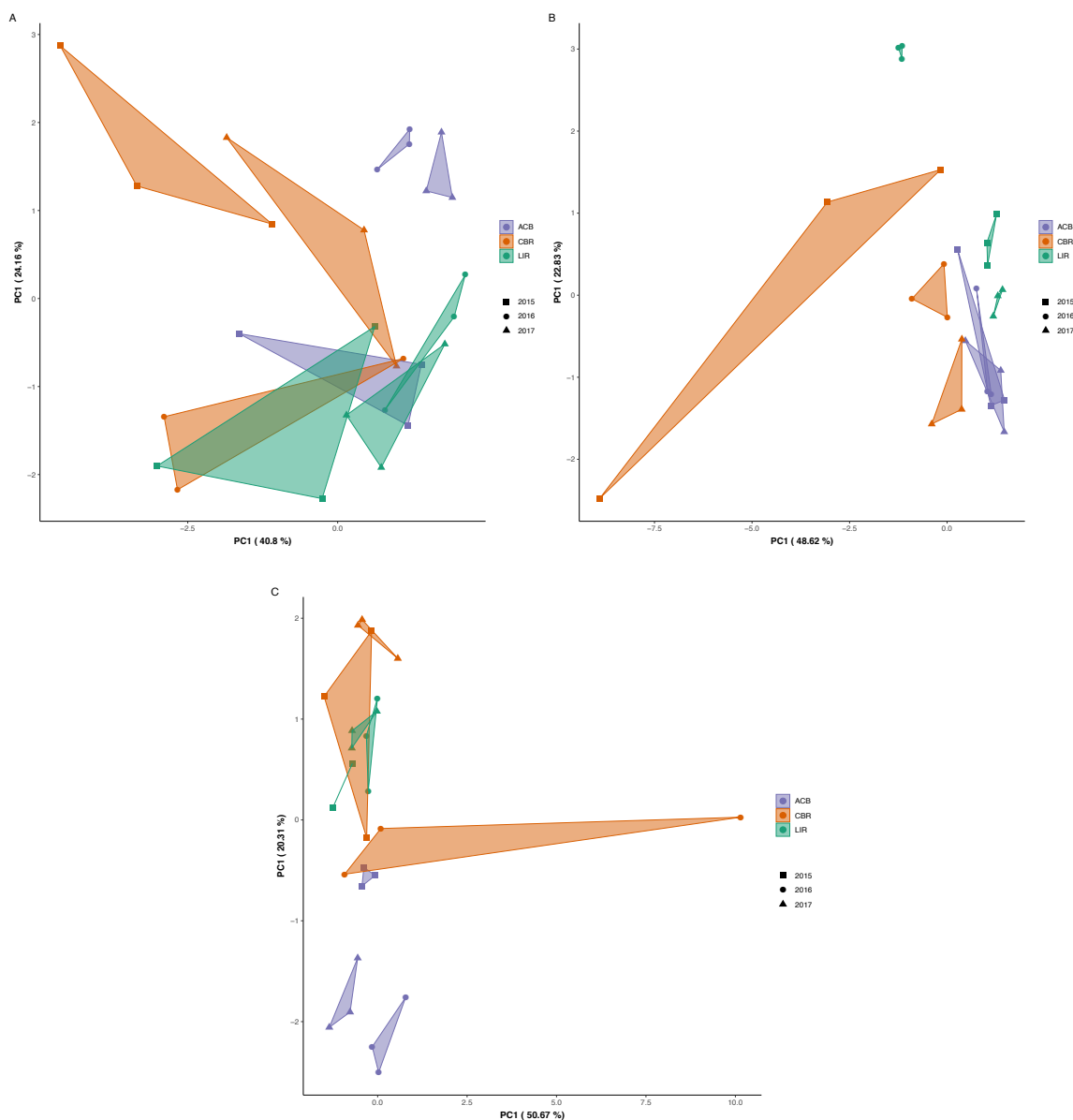


Figure 4: PCA of water chemistry parameters from grab samples collected once seasonally at each station. Due to known seasonal variation in water chemistry parameters, PCAs were blocked by season: Spring (A), Summer (B), and Fall (C). Analytes include dissolved and particulate nitrogen and phosphate, as well as sestonic chlorophyll concentration and total suspended solids. See Table 1 for a full listing of analytes included.

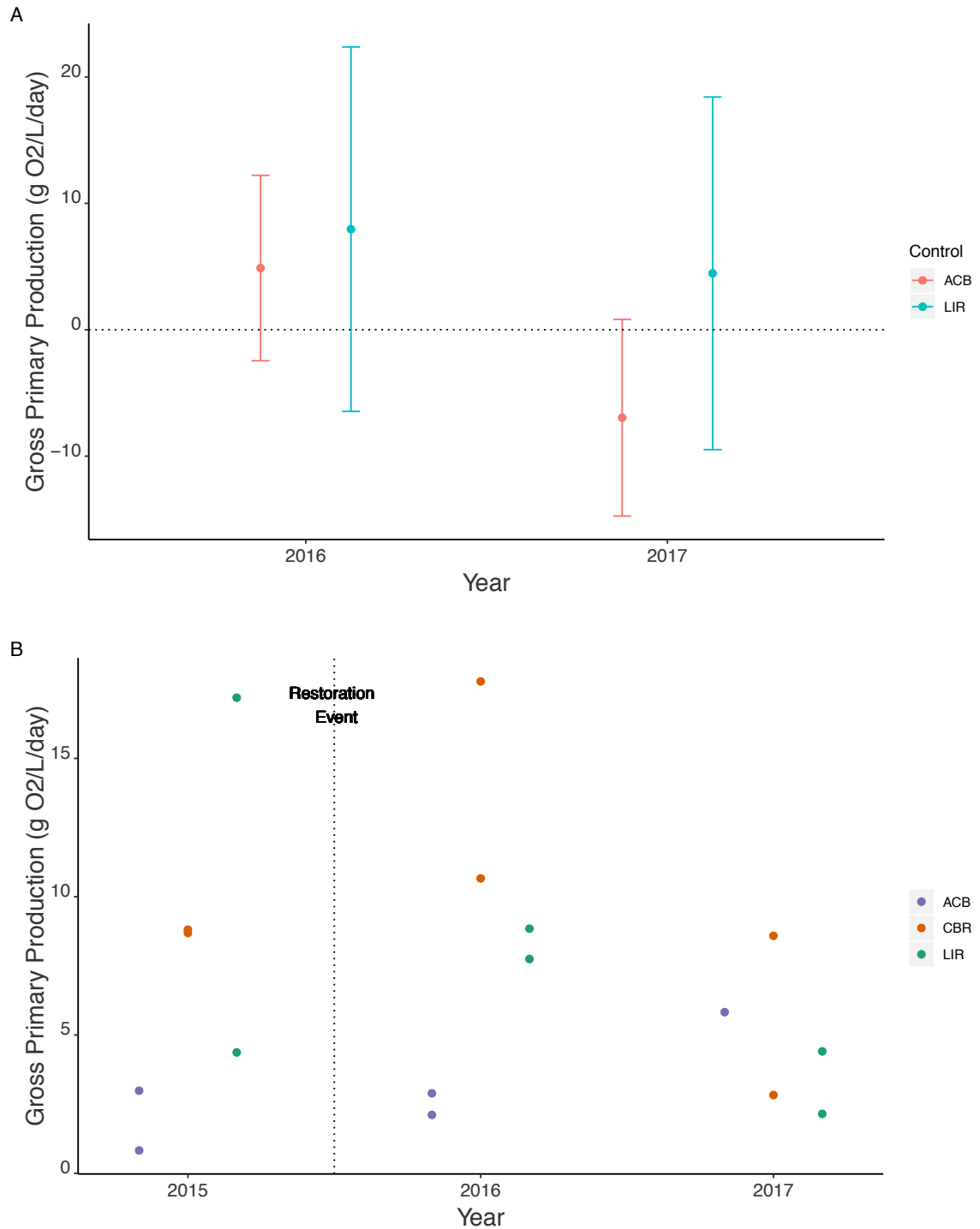


Figure 5: Summer gross primary production as given by BACI effect size (A) and actual values (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests ($\alpha=0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

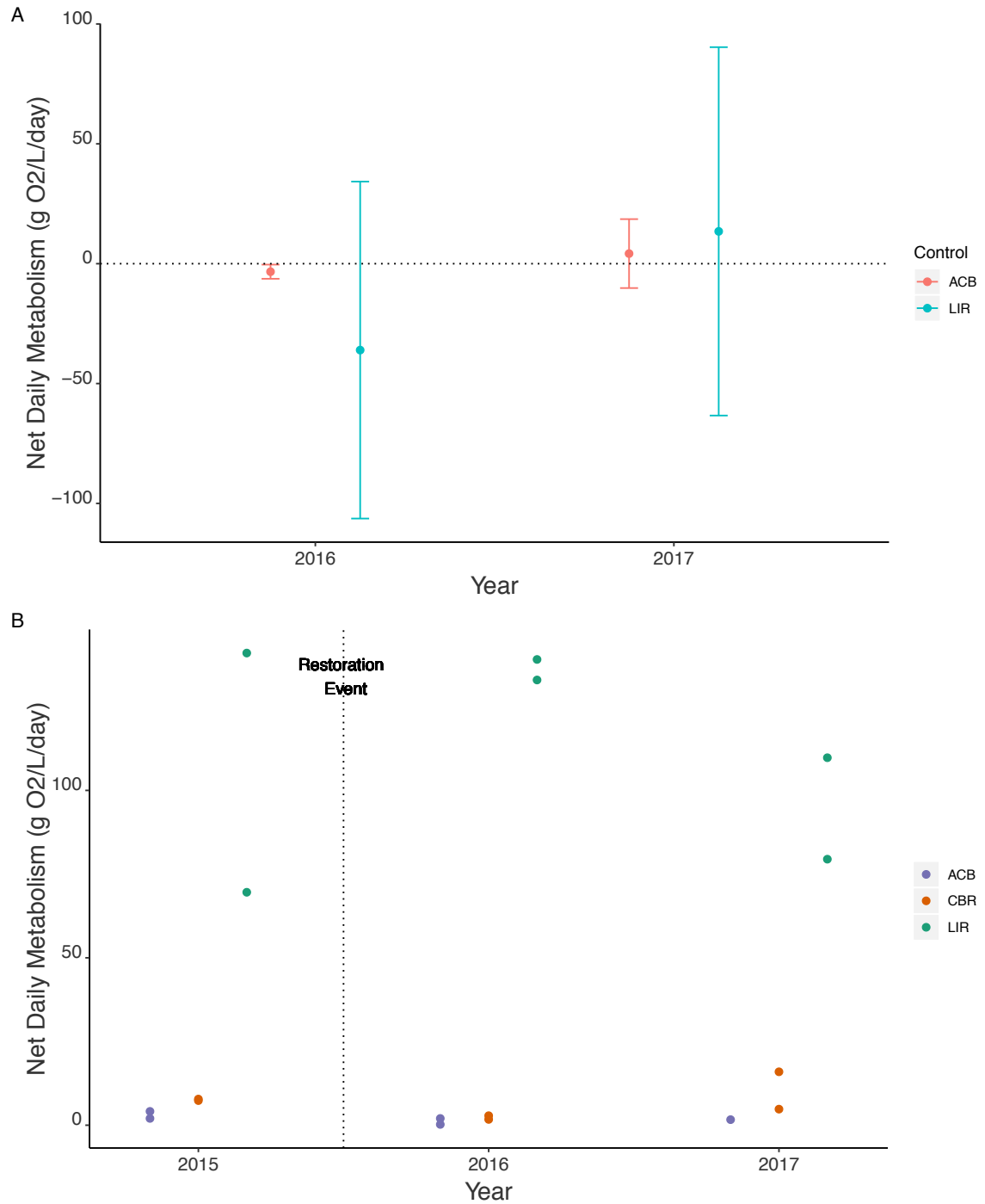


Figure 6: Summer net daily metabolism as given by BACI effect size (A) and actual values (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests ($\alpha = 0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

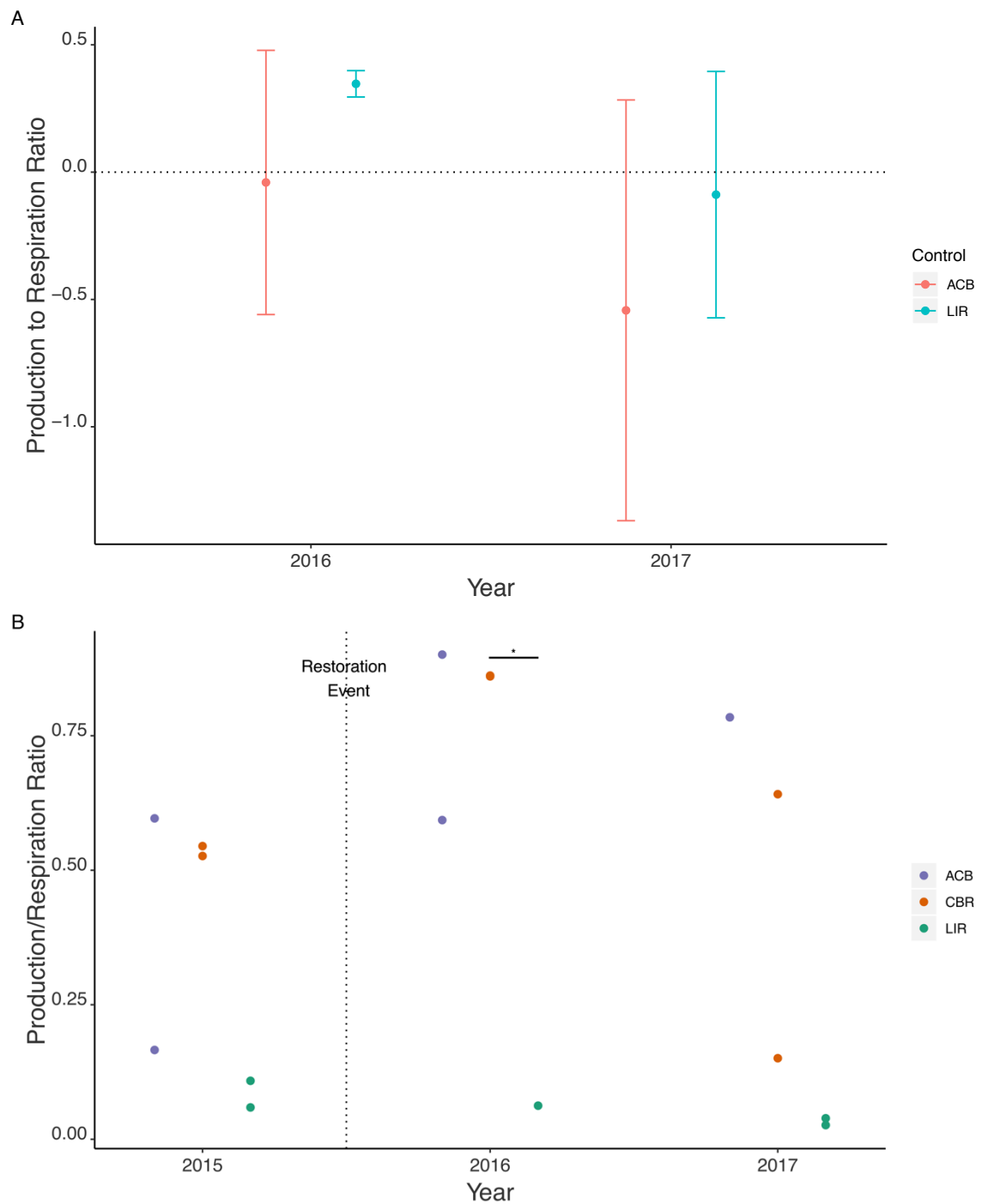


Figure 7: Summer production to respiration ratio as given by BACI effect size (A) and actual values (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests ($\alpha = 0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

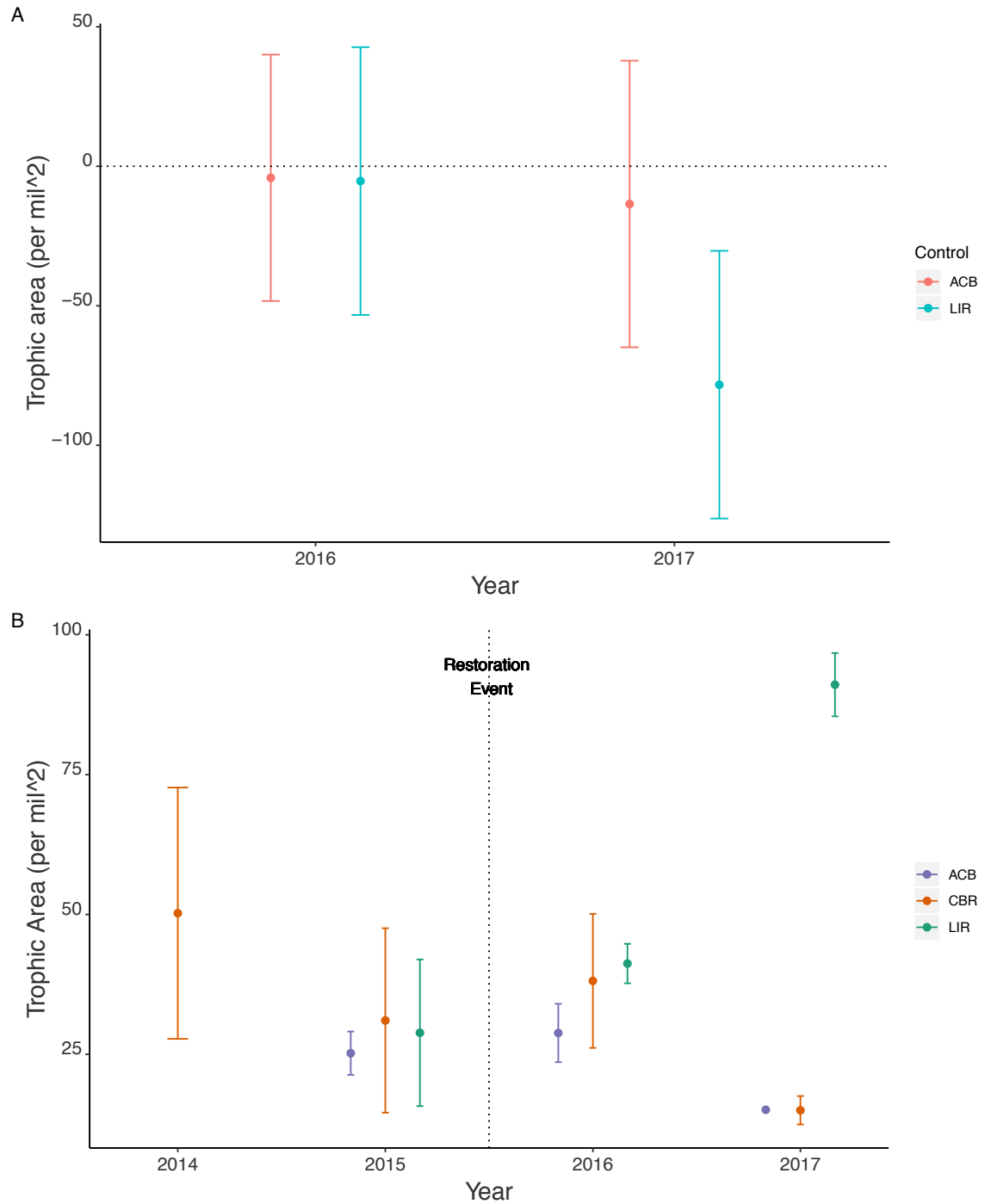


Figure 8: Isotopic value derived trophic area as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests (Bonferonni-adjusted $\alpha=0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

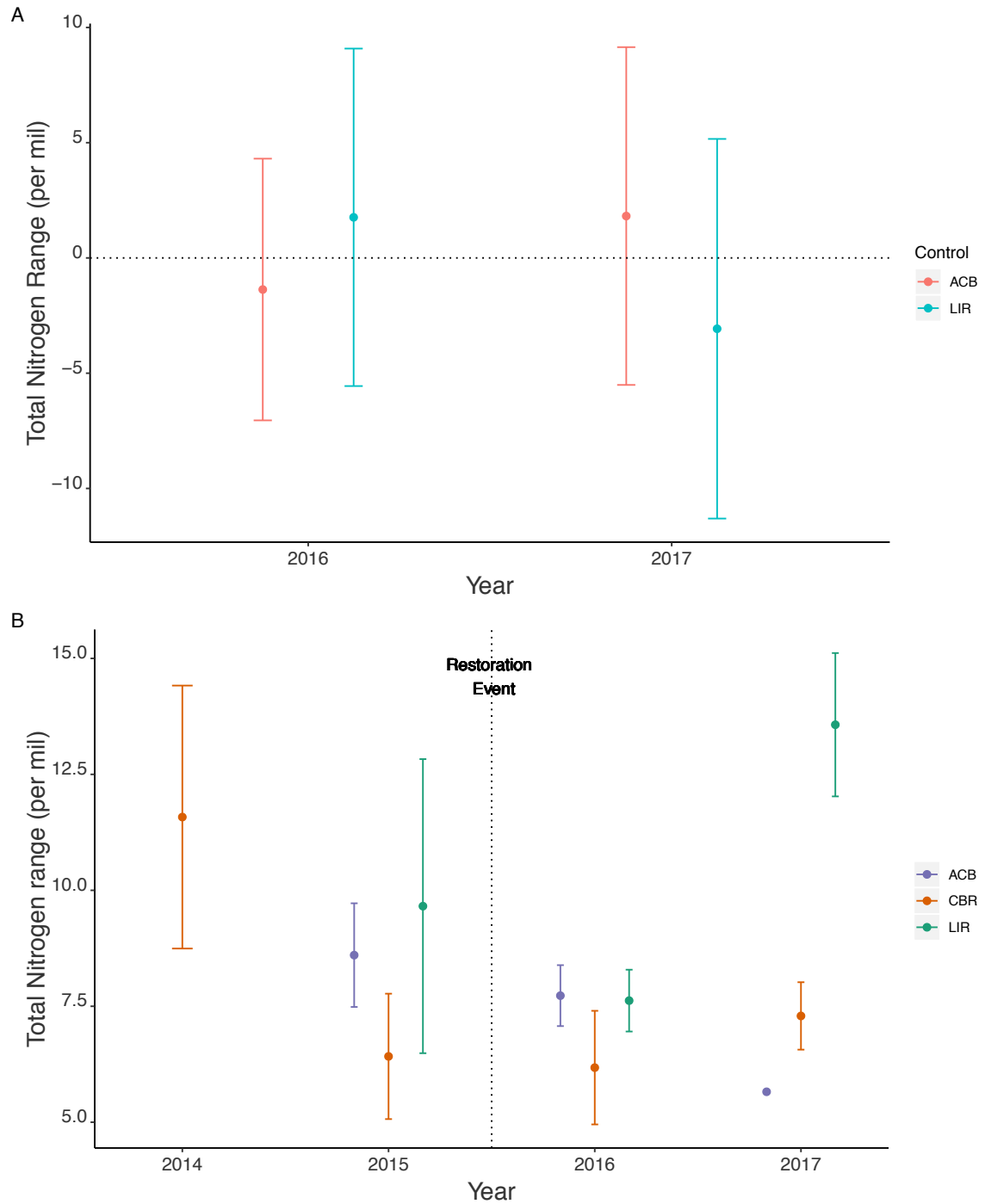


Figure 9: Isotopic value derived total nitrogen range as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests (Bonferonni-adjusted $\alpha = 0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

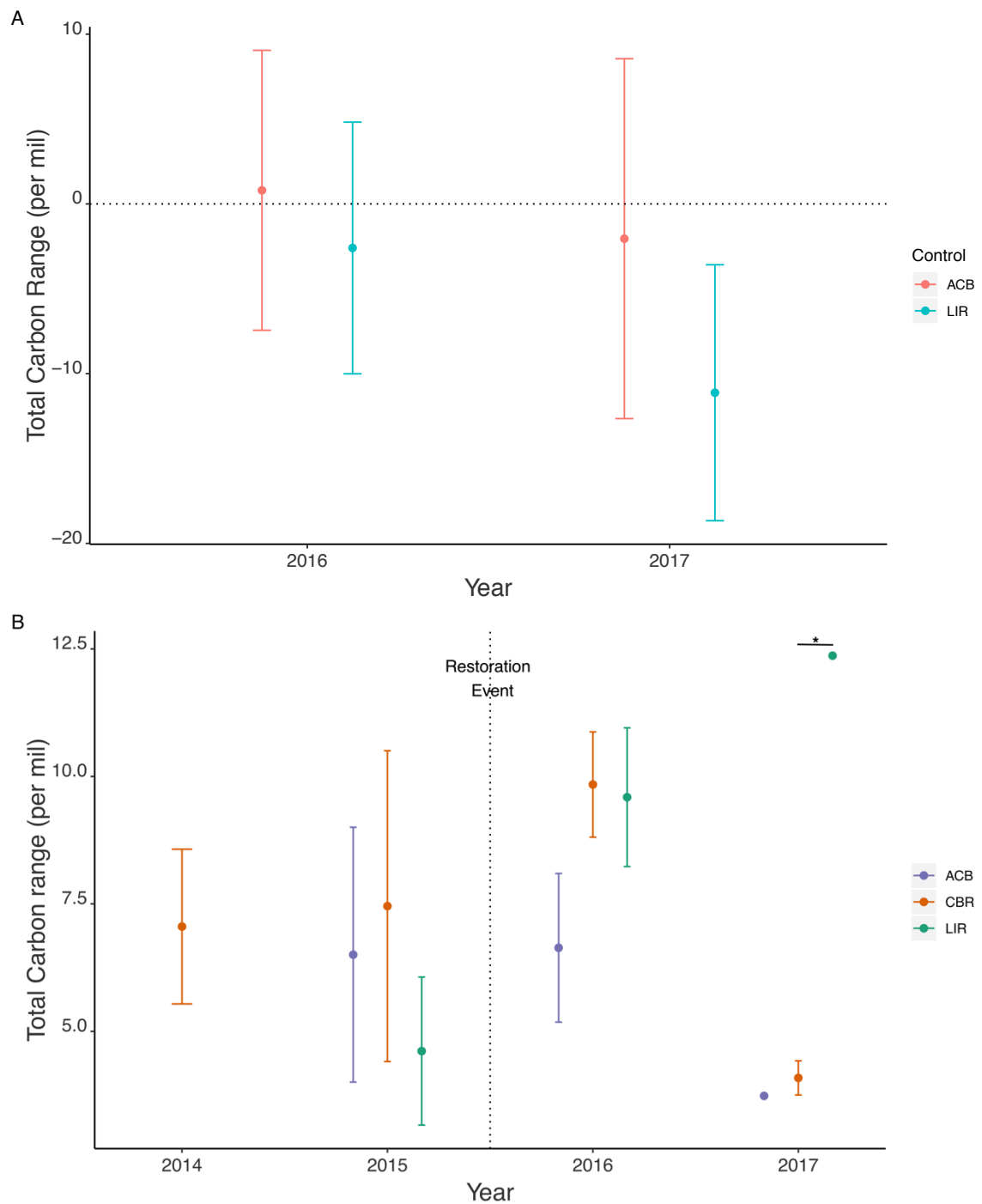


Figure 10: Isotopic value derived total carbon range as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch’s t-tests (Bonferonni-adjusted $\alpha = 0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

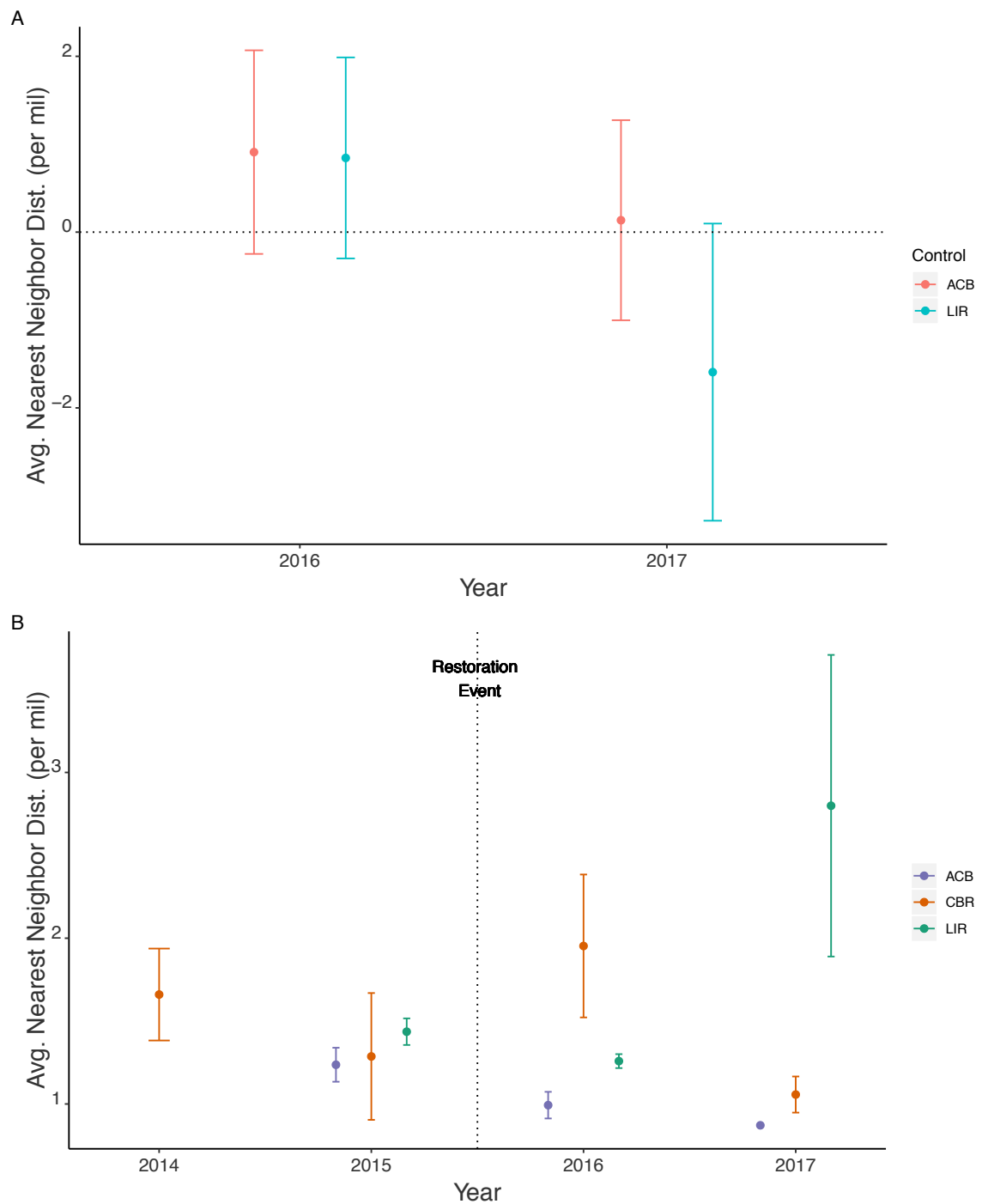


Figure 11: Isotopic value derived average nearest neighbor distance as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests (Bonferonni-adjusted $\alpha = 0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

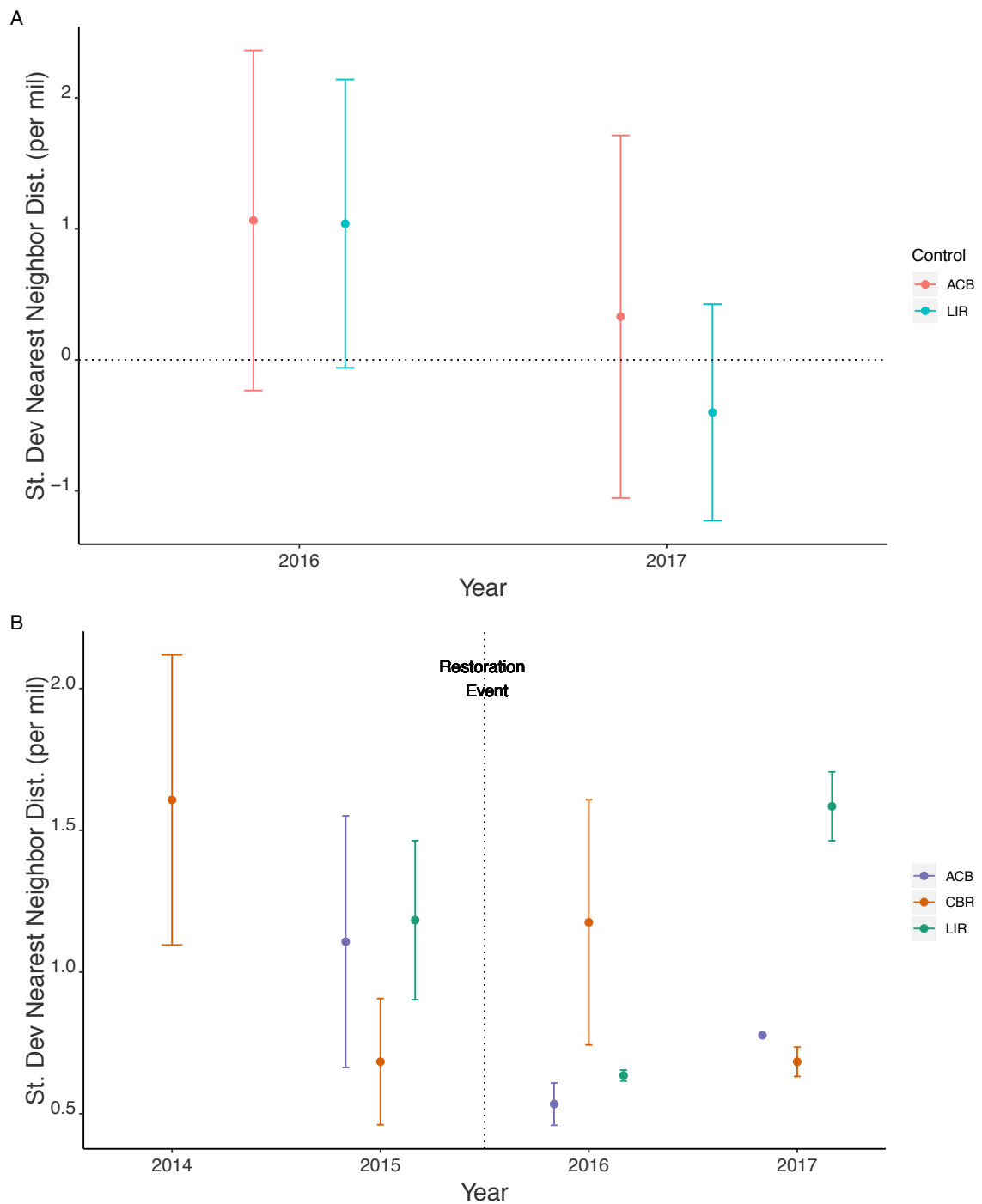


Figure 12: Isotopic value derived standard deviation of nearest neighbor distances as given by BACI effect size (A) and actual values with standard error (B). Significant differences are noted by confidence interval range not overlapping zero (A) or pair-wise Welch's t-tests (Bonferonni-adjusted $\alpha = 0.008$) between CBR and other treatments within each year (B), marked with (*). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

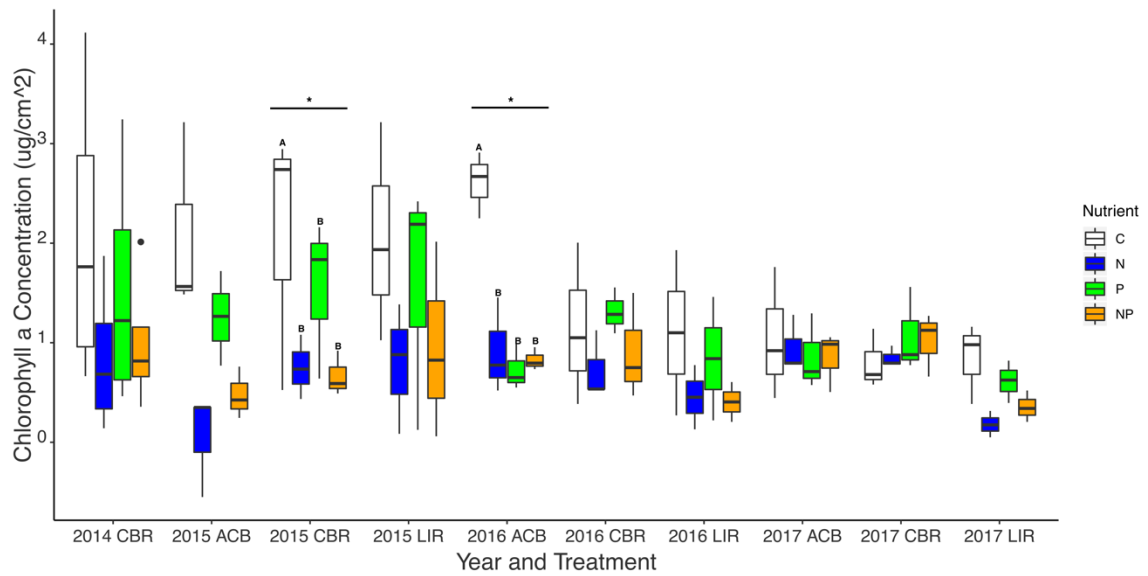


Figure 13: Nutrient limitation experiments for each treatment and year combination, graphed as total chlorophyll present after 3 week incubation. Within each year/treatment, ANOVA was performed to determine if growth was significantly different in any of the growth conditions. Significant differences ($\alpha=0.05$) within each year/treatment are marked with (*). Tukey's HSD post hoc differences are marked with distinct letters within each ANOVA. ACB= Active Cranberry Bog, CBR= Cranberry Bog Restoration, LIR = Least Impacted Reference; C= Control, N= Nitrogen Addition, P= Phosphate Addition, NP = Nitrogen and Phosphate Addition.

APPENDIX A

Table A.1: Effect size (BACI) of change before and after restoration (2015 to 2016, or 2015 to 2017) relative to control (LIR or ACB) treatment. Each effect size is calculated as described in-text, with 95% confidence intervals generated from pooled variance. Bolding represents effects with confidence intervals not overlapping 0. ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

Table A.2: Within year differences between CBR and control treatments. T values represent Welch's two-tailed t-tests between CBR and listed treatment in the same year. Due to unequal variances, Welch's t-tests calculate v, a modified df. Bolding represents changes that are significantly different ($\alpha = 0.008$, Bonferroni correction for 6 comparisons per year). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

Table A.1

Year	Control	Gross Primary Productivity			Net Daily Metabolism			P to R Ratio	
		d	95% CI		d	95% CI		d	95% CI
2016	LIR	7.960	[-6.457, 22.376]		-36.039	[-106.326, 34.248]		0.347	[0.295, 0.399]
2016	ACB	4.879	[-2.457, 12.215]		-3.339	[-6.296, -0.382]		-0.040	[-0.559, 0.478]
2017	LIR	4.463	[-9.49, 18.415]		13.488	[-63.353, 90.328]		-0.088	[-0.572, 0.395]
2017	ACB	-6.959	[-14.74, 0.821]		4.214	[-10.162, 18.589]		-0.543	[-1.369, 0.283]

Table A.2

Year	CBR:Treatment	Gross Primary Productivity			Net Daily Metabolism			Production to Respiration Ratio		
		T value	P value	v	T value	P value	v	T value	P value	v
2015	ACB	6.312	0.099	1.006	-4.177	0.134	1.091	0.717	0.604	1.004
	LIR	-0.317	0.804	1.000	2.735	0.223	1.000	17.117	0.019	1.272
2016	ACB	3.271	0.184	1.024	-1.102	0.404	1.674	0.741	0.594	1.000
	LIR	1.645	0.339	1.048	42.941	0.012	1.066	1107.659	<0.001	1.001
2017	ACB	-	-	-	-	-	-	-	-	-
	LIR	0.786	0.550	1.301	5.217	0.083	1.266	1.480	0.378	1.001

APPENDIX B

Table B.1: Effect size (BACI) of change before and after restoration (2015 to 2016, or 2015 to 2017) relative to control (LIR or ACB) treatment. Each effect size is calculated as described in-text, with 95% confidence intervals generated from pooled variance. Bolding represents effects with confidence intervals not overlapping 0. ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

Table B.2: Within year differences between CBR and control treatments. T values represent Welsh's two-tailed t-tests between CBR and listed treatment in the same year. Due to unequal variances, Welsh's t-tests calculate v, a modified df. Bolding represents changes that are significantly different ($\alpha = 0.008$, Bonferroni correction for 6 comparisons per year). ACB = Active Cranberry Bog, CBR = Cranberry Bog Restoration, LIR = Least Impacted Reference.

Table B.1

Year	Control	Trophic Area		Nitrogen Range		Carbon Range	
		d	95% CI	d	95% CI	d	95% CI
2016	LIR	-5.299	[-53.296, 42.697]	-2.594	[-10.009, 4.82]	1.762	[-5.558, 9.083]
2016	ACB	-4.124	[-48.288, 40.039]	0.801	[-7.445, 9.048]	-1.370	[-7.047, 4.307]
2017	LIR	-78.292	[-126.272, -30.311]	-11.124	[-18.664, -3.585]	-3.072	[-11.307, 5.163]
2017	ACB	-13.521	[-64.882, 37.84]	-2.049	[-12.649, 8.551]	1.818	[-5.509, 9.144]

Table B.1 continued

Year	Control	Mean NND		St. Dev of NND	
		d	95% CI	d	95% CI
2016	LIR	0.844	[-0.3, 1.987]	1.040	[-0.061, 2.141]
2016	ACB	0.910	[-0.248, 2.068]	1.064	[-0.234, 2.363]
2017	LIR	-1.593	[-3.284, 0.097]	-0.402	[-1.228, 0.425]
2017	ACB	0.135	[-1.003, 1.274]	0.330	[-1.055, 1.714]

Table B.2

Year	CBR:Treatment	Trophic Area		Nitrogen Range		Carbon Range	
		T value	P value	v	T value	P value	v
2015	ACB	0.732	0.519	2.905	-0.072	0.947	3.366
	LIR	0.105	0.922	3.806	-0.926	0.430	2.696
2016	ACB	0.713	0.532	2.733	-1.117	0.344	3.063
	LIR	-0.248	0.824	2.344	-1.036	0.374	3.089
2017	ACB	-	-	-	-	-	-
	LIR	-10.308	0.035	1.275	-3.097	0.150	1.303
						-24.612	0.002
							2.025

Table B.2 continued

Table B.2 continued							
Year	CBR:Treatment	Mean NND			St. Dev of NND		
		T value	P value	v	T value	P value	v
2015	ACB	0.127	0.910	2.286	-0.852	0.458	2.946
	LIR	-0.382	0.736	2.175	-1.394	0.239	3.084
2016	ACB	2.189	0.152	2.139	1.462	0.275	2.120
	LIR	1.603	0.248	2.038	1.249	0.338	2.008
2017	ACB	-	-	-	-	-	-
	LIR	-1.556	0.360	1.019	-5.718	0.076	1.249

LITERATURE CITED

- Angermeier, P.L. (1995). Ecological Attributes of Extinction-Prone Species: Loss of Freshwater Fishes of Virginia. *Conserv. Biol.* 9, 143–158.
- Bales, J.D., and Nardi, M.R. (2007). Automated Routines for Calculating Whole-Stream Metabolism: Theoretical Background and User's Guide. USGS.
- Barbour, M.T., Gerritsen, J., Snyder, B.D., and Stribling, J.B. (1999). Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, (Washington, D.C.: U.S. Environmental Protection Agency; Office of Water).
- Barton, B.A. (1977). Short-term effects of highway construction on the limnology of a small stream in southern Ontario*. *Freshw. Biol.* 7, 99–108.
- Beisner, B.E., Haydon, D.T., and Cuddington, K. (2003). Alternative Stable States in Ecology. *Front. Ecol. Environ.* 1, 376–382.
- Bunn, S.E., and Arthington, A.H. (2002). Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environ. Manage.* 30, 492–507.
- Carpenter, S.R., and Brock, W.A. (2006). Rising variance: a leading indicator of ecological transition. *Ecol. Lett.* 9, 311–318.
- Christie, A.P., Amano, T., Martin, P.A., Shackelford, G.E., Simmons, B.I., and Sutherland, W.J. (2019). Simple study designs in ecology produce inaccurate estimates of biodiversity responses. *J. Appl. Ecol.* 56, 2742–2754.
- Elser, J.J., Bracken, M.E.S., Cleland, E.E., Gruner, D.S., Harpole, W.S., Hillebrand, H., Ngai, J.T., Seabloom, E.W., Shurin, J.B., and Smith, J.E. (2007). Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. *Ecol. Lett.* 10, 1135–1142.
- ESRI ArcGIS 10.4.
- Evju, M., Hagen, D., Kyrkjeeide, M.O., and Köhler, B. (2020). Learning from scientific literature: Can indicators for measuring success be standardized in “on the ground” restoration? *Restor. Ecol.* 28, 519–531.
- Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., and Holling, C.S. (2004). Regime Shifts, Resilience, and Biodiversity in Ecosystem Management. *Annu. Rev. Ecol. Evol. Syst.* 35, 557–581.
- Franck, E. (2017). Establishment of Baseline Physical, Chemical, and Biological Stream Conditions at a Passive Cranberry Bog Restoration. Grad. Masters Theses.

- Gabriele, W., Welty, N., and Hein, T. (2013). Limitations of stream restoration for nitrogen retention in agricultural headwater streams. *Ecol. Eng.* 60, 224–234.
- Gibeau, G.G., and Miller, M.C. (1989). A Micro-Bioassay for Epilithon using Nutrient-Diffusing Artificial Substrata. *J. Freshw. Ecol.* 5, 171–176.
- Guecker, B., Boechat, I.G., and Giani, A. (2009). Impacts of agricultural land use on ecosystem structure and whole-stream metabolism of tropical Cerrado streams. *Freshw. Biol.* 54, 2069–2085.
- Harvey, B.C., White, J.L., and Nakamoto, R.J. (2009). The effect of deposited fine sediment on summer survival and growth of rainbow trout in riffles of a small stream. *North Am. J. Fish. Manag.* 29, 434–440.
- Hauer, F.R., and Lamberti, G.A. (2006). *Methods in Stream Ecology* (Academic Press).
- Hewitt JE, and Thrush SF (2010). Empirical evidence of an approaching alternate state produced by intrinsic community dynamics, climatic variability and management actions. *Mar. Ecol. Prog. Ser.* 413, 267–276.
- Hogsden, K.L., and Harding, J.S. (2014). Isotopic metrics as a tool for assessing the effects of mine pollution on stream food webs. *Ecol. Indic.* 36, 339–347.
- Holling, C.S. (1996). Surprise for Science, Resilience for Ecosystems, and Incentives for People. *Ecol. Appl.* 6, 733–735.
- Izagirre, O., Agirre, U., Bermejo, M., Pozo, J., and Elosegui, A. (2008). Environmental controls of whole-stream metabolism identified from continuous monitoring of Basque streams. *J. North Am. Benthol. Soc.* 27, 252–268.
- Kauffman, J.B., Beschta, R.L., Otting, N., and Lytjen, D. (1997). An Ecological Perspective of Riparian and Stream Restoration in the Western United States. *Fisheries* 22, 12–24.
- Kimmel, W.G., and Argent, D.G. (2006). Efficacy of two-pass electrofishing employing multiple units to assess stream fish species richness. *Fish. Res.* 82, 14–18.
- Layman, C.A., Arrington, D.A., Montaña, C.G., and Post, D.M. (2007). Can Stable Isotope Ratios Provide for Community-Wide Measures of Trophic Structure? *Ecology* 88, 42–48.
- Legendre, P., and Legendre, L. (2012). Complex ecological data sets. In *Developments in Environmental Modelling*, (Elsevier), pp. 1–57.
- Madsen, J.D., and Adams, M.S. (1989). The distribution of submerged aquatic macrophyte biomass in a eutrophic stream, Badfish Creek: the effect of environment. *Hydrobiologia* 171, 111–119.

- Mattson, M.D., Godfrey, P.J., Barletta, R.A., and Aiello, A. (2004). Eutrophication and Aquatic Plant Management in Massachusetts: Final Generic Environmental Impact Report.
- McCain, M., Fuller, D., Decker, L., and Overton, K. (1990). Stream habitat classification and inventory procedures for northern California. USDA For. Serv. Pac. Southwest Reg. FHR Curr. San Franc. CA 94111 1–16.
- Newbold, J.D., O'Neill, R.V., Elwood, J.W., and Van Winkle, W. (1982). Nutrient Spiralling in Streams: Implications for Nutrient Limitation and Invertebrate Activity. *Am. Nat.* 120, 628–652.
- Palmer, M.A., Hondula, K.L., and Koch, B.J. (2014). Ecological Restoration of Streams and Rivers: Shifting Strategies and Shifting Goals. *Annu. Rev. Ecol. Evol. Syst.* 45, 247–269.
- Poff, N.L. (1997). Landscape Filters and Species Traits: Towards Mechanistic Understanding and Prediction in Stream Ecology. *J. North Am. Benthol. Soc.* 16, 391–409.
- Power, M.E., Parker, M.S., and Wooten, J.T. (1995). Disturbance and food chain length in rivers. In *Food Webs: Integration of Patterns and Dynamics*, G.A. Polis, and K.O. Winemiller, eds. (New York: Chapman Hall), p.
- R Core Team (2018). R: A Language and Environment for Statistical Computing (Vienna, Austria: R Foundation for Statistical Computing).
- Roedenbeck, I., Fahrig, L., Findlay, C.S., Houlihan, J., Jaeger, J., Klar, N., Kramer-Schadt, S., and van der Grift, E. (2007). The Rauischholzhausen Agenda for Road Ecology. *Ecol. Soc.* 12.
- Sefick, Jr., S. (2016). StreamMetabolizer - A package for calculating single station metabolism from diurnal oxygen curves.
- Stanley, E.H., Powers, S.M., and Lottig, N.R. (2010). The evolving legacy of disturbance in stream ecology: concepts, contributions, and coming challenges. *J. North Am. Benthol. Soc.* 29, 67–83.
- Stotz, G.C., Gianoli, E., Patchell, M.J., and Cahill, J.F. (2017). Differential responses of native and exotic plant species to an invasive grass are driven by variation in biotic and abiotic factors. *J. Veg. Sci.* 28, 325–336.
- Sudduth, E.B., Hassett, B.A., Cada, P., and Bernhardt, E.S. (2011). Testing the Field of Dreams Hypothesis: functional responses to urbanization and restoration in stream ecosystems. *Ecol. Appl.* 21, 1972–1988.

- Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., and Cushing, C.E. (1980). The River Continuum Concept. *Can. J. Fish. Aquat. Sci.* 37, 130–137.
- Villéger, S., Miranda, J.R., Hernández, D.F., and Mouillot, D. (2010). Contrasting changes in taxonomic vs. functional diversity of tropical fish communities after habitat degradation. *Ecol. Appl.* 20, 1512–1522.
- Wohl, E., Angermeier, P.L., Bledsoe, B., Kondolf, G.M., MacDonnell, L., Merritt, D.M., Palmer, M.A., Poff, N.L., and Tarboton, D. (2005). River restoration. *Water Resour. Res.* 41, W10301.
- Wold, A.P., and Hershey, A.E. (1999). Spatial and Temporal Variability of Nutrient Limitation in 6 North Shore Tributaries to Lake Superior. *J. North Am. Benthol. Soc.* 18, 2–14.
- (1998). Standard Methods for the Examination of Water and Wastewater (American Public Health Association).

CHAPTER 5

CONCLUSION

OVERVIEW

The goal of this study was to investigate the changes to taxonomic and ecosystem level biodiversity within the first two years of an active restoration of a flow-through cranberry bog (CBR). Although limited in temporal scale, the use of a broad slate of response variables enabled us to identify early changes to the system and provided a weight-of-evidence approach to state shifts even if equilibrium is not achieved within the duration of the study. Therefore, this research utilized a known restoration project to provide Before-After-Control-Impact (BACI) experiment in disturbance's effect on habitat, freshwater macroinvertebrate assemblage, and ecosystem process across multiple scales and through the three major attributes of biodiversity. Two control systems were used, a least-impacted reference condition (LIR), with no history of agriculture, and an actively farmed cranberry bog (ACB). Overall, I expected to find that the cranberry bog restoration would result in a first-year response similar to that expected from a classic pulse disturbance. However, given the changes to the landscape and processes brought about by active restoration, I expected the "recovery" from that disturbance to trend toward an alternate state dissimilar to either its initial state or that of an active cranberry bog.

However, the majority of my findings suggest that two years post-restoration, the system returned to largely similar to its pre-restoration state, suggesting a pulse disturbance without subsequent state shift.

SUMMARY OF FINDINGS

My first objective was to quantify changes in watershed scale land use and reach scale in-stream physical habitat over the course of the restoration. I found that all three treatments were relatively similar in watershed scale composition and natural land use fragmentation, allowing both LIR and ACB to be used as BACI controls, normalizing background variability in analysis of changes at the restoration site. Secondly, I found that the scope of the restoration project within its coastal watershed had the potential to alter nearly 10% of the agricultural land use at a landscape level and reduce natural land use fragmentation. This level of watershed composition change cannot be ignored as a potential driver of on-site changes, as many other studies have shown watershed land use impacting biologic impacts (Allan, 2004), particularly with respect to agriculture (Atkinson et al., 2014) and urbanization (Price et al., 2006). Finally, local scale habitat alterations following restoration were broadly not found to be persistent after 2 years, either through bulk analysis using principle component analysis (PCA) or through detailed analysis of their contribution to overall habitat heterogeneity. This contrasted with my expectation of compositional change to habitat following active rechannelization and introduction of woody debris. Although longer-term analysis of habitat changes are required to assess the

restoration's ultimate impact on in-stream habitat, the preliminary lack of overall change may have some bearing on expectations of alterations to the biologic community.

As a second objective, I examined the compositional and functional components of the freshwater macroinvertebrate assemblages at all three treatments. I found that for univariate diversity measures, all three treatments had largely non-significant differences related to family richness or Shannon's Diversity. When compared for taxonomic composition with NMDS however, CBR showed an initial similarity to the ACB sites, despite the five years of passive restoration at CBR. This contrasts with expectations of change in the absence of active management (Wohl et al., 2005), but does match other studies which suggest the legacy effects of agriculture result in similar community composition (Engel, 2010; Harding et al., 1998). Over the course of the restoration, CBR exhibited a shift in community composition away from its pre-restoration state in the first year following restoration, but by year two, it had become compositionally similar to the initial state. This suggests a similar community response to that predicted by a pulse disturbance (Suding et al., 2004). When examined the assemblage functionally through feeding groups, the same pattern emerges, both driven by the relative increase in one lentic specializing herbivorous family, the *Corixidae*. As a result, although simple diversity metrics showed no significant differences between sites, multi-dimensional analysis of the assemblages showed patterns consistent with a pulse disturbance.

Finally, my last objective examined stream processes through water column chemistry, limiting nutrients, stream metabolism, and functional food web metrics. I found that water chemistry was not consistently changed following restoration, was highly seasonal, and the differences between treatments and years was driven primarily by

dissolved nutrients, rather than particulates or chlorophyll content. Limiting nutrients were not found for any site and did not change over the course of the study. This may be due to the lake source water of all three treatments and the high levels of eutrophication present in Southeastern Massachusetts (Mattson et al., 2004). Stream metabolism metrics saw large swings in 2016 for CBR related to increased production and decline in net metabolism, both not maintained in 2017. This suggests a brief period of metabolism shift that coincides with the dredging of new channel, reducing sand coverage, resuspending nutrients, and allowing for macrophyte propagule resuspension and growth, which boost in-stream production (Madsen and Adams, 1989). Finally, no change was observed over the course of the study for functional food web metrics, which suggests no notable changes to food web length, carbon inputs, or niche redundancy. As the largest source of potential change is leaf input from riparian canopy (Sudduth et al., 2011) and introduction of new top predators (Power et al., 1995), these food web metrics may require longer recovery to show variability. The lack of significant change in these functional measures is not entirely surprising (Palmer et al., 2014) but does prompt questions about how they may alter over the next decade of recovery.

Taken together, these findings suggest that the first two years of the restoration most closely follow a pulse disturbance, where the initial state is resumed after a one year perturbation caused by the activities of the restoration. This finding is not surprising in the short term, although the weight-of-evidence approach and broad range of response variables were chosen to provide the greatest ability to detect early shifts (Wohl et al., 2005). Key shifts required by process-based restoration require integration of the stream community with hydromorphic and riparian processes, which may require longer periods

to assess changes (Bakke et al., 2020; McMillan et al., 2014). Restoration projects are often assessed over the longer term, and even then functional change is rarely demonstrated (Palmer et al., 2014). Thus, over the near-term, a pulse disturbance response is to be expected for functional change, but is surprising for structural and compositional measures, especially those related to hydrologic connectivity and in-stream habitat, around which restorative activities were based.

RECOMMENDED FUTURE WORK

Active restoration is a two-part process, with an initial disturbance phase to eliminate the anthropogenic impairments to the system, and a longer gradual recovery as natural succession shifts the now unmanaged system into a state ideally similar to targeted reference conditions. The scope of this study is limited to the initial disturbance phase of restoration, and as such its conclusions in turn yield questions about the recovery phase. There is companion study in our research group investigating habitat diversity and fish compositional, structural, and functional biodiversity over the same time period that may also provide additional insight into the CBR restoration disturbance. The results of this study have two complementary natural follow-ups. The first is to continue evaluation of post-restoration biodiversity at CBR over the next decade, as altered flow conditions and riparian regrowth, including canopy shading and leaf input, play a role in altering the habitat, macroinvertebrate assemblage, and stream processes. The second is the comparison of new restoration projects to the findings of this study, to determine if these patterns are site specific or relate to larger phases of the restoration process. The

participation of CBR under the umbrella organization Living Observatory (LO), provides real potential to pursue both of these next steps. LO is a “public interest learning collaborative” of researchers interested in pursuing questions about restoration and specific impacts on the aquatic, terrestrial, and microbial systems of wetlands (Living Observatory 2020). In addition to the maintenance of long-term research at CBR offered by its LO membership, additional restoration projects conducted by the Massachusetts Division of Ecological Restoration have been proposed to be adopted into the LO system, creating the potential to use space-for-time substitutions to ask questions about the trajectory of cranberry bog restoration over longer time scales.

LITERATURE CITED

- Allan, J.D. (2004). Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems. *Annu. Rev. Ecol. Evol. Syst.* 35, 257–284.
- Atkinson, C.L., Christian, A.D., Spooner, D.E., and Vaughn, C.C. (2014). Long-lived organisms provide an integrative footprint of agricultural land use. *Ecol. Appl.* 24, 375–384.
- Bakke, P.D., Hrachovec, M., and Lynch, K.D. (2020). Hyporheic Process Restoration: Design and Performance of an Engineered Streambed. *Water* 12, 425.
- Engel, H. (2010). Stream quality among active and restoring river-based cranberry bogs. Unpubl. MBL Woods Hole.
- Harding, J.S., Benfield, E.F., Bolstad, P.V., Helfman, G.S., and Jones, E.B.D. (1998). Stream biodiversity: The ghost of land use past. *Proc. Natl. Acad. Sci. U. S. A.* 95, 14843–14847.
- Madsen, J.D., and Adams, M.S. (1989). The distribution of submerged aquatic macrophyte biomass in a eutrophic stream, Badfish Creek: the effect of environment. *Hydrobiologia* 171, 111–119.
- Mattson, M.D., Godfrey, P.J., Barletta, R.A., and Aiello, A. (2004). Eutrophication and Aquatic Plant Management in Massachusetts: Final Generic Environmental Impact Report.
- McMillan, S.K., Tuttle, A.K., Jennings, G.D., and Gardner, A. (2014). Influence of Restoration Age and Riparian Vegetation on Reach-Scale Nutrient Retention in Restored Urban Streams. *JAWRA J. Am. Water Resour. Assoc.* 50, 626–638.
- Palmer, M.A., Hondula, K.L., and Koch, B.J. (2014). Ecological Restoration of Streams and Rivers: Shifting Strategies and Shifting Goals. *Annu. Rev. Ecol. Evol. Syst.* 45, 247–269.
- Power, M.E., Parker, M.S., and Wooten, J.T. (1995). Disturbance and food chain length in rivers. In *Food Webs: Integration of Patterns and Dynamics*, G.A. Polis, and K.O. Winemiller, eds. (New York: Chapman Hall), p.
- Price, S.J., Dorcas, M.E., Gallant, A.L., Klaver, R.W., and Willson, J.D. (2006). Three decades of urbanization: Estimating the impact of land-cover change on stream salamander populations. *Biol. Conserv.* 133, 436–441.
- Sudduth, E.B., Hassett, B.A., Cada, P., and Bernhardt, E.S. (2011). Testing the Field of Dreams Hypothesis: functional responses to urbanization and restoration in stream ecosystems. *Ecol. Appl.* 21, 1972–1988.

- Suding, K.N., Gross, K.L., and Houseman, G.R. (2004). Alternative states and positive feedbacks in restoration ecology. *Trends Ecol. Evol.* *19*, 46–53.
- Wohl, E., Angermeier, P.L., Bledsoe, B., Kondolf, G.M., MacDonnell, L., Merritt, D.M., Palmer, M.A., Poff, N.L., and Tarboton, D. (2005). River restoration. *Water Resour. Res.* *41*, W10301.